Appendix L

System Assessment Capability:
A 10,000-Year, Post-Closure Assessment

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L.1 Introduction

In late 1997, the U.S. Department of Energy (DOE) established the Groundwater/Vadose Zone Integration Project with Bechtel Hanford, Inc. (BHI), the Hanford Site Environmental Restoration Contractor, as manager. The project transitioned to Fluor Hanford, the Project Hanford Management Contractor, in July 2002, and has been renamed the Groundwater Protection Program. Pacific Northwest National Laboratory (PNNL) is a partner in the project. The mission of the project is to coordinate and integrate projects that characterize, monitor, and clean up contaminants in the groundwater and vadose zone (the soil between the ground surface and the groundwater) beneath the Hanford Site. The Groundwater Protection Project also incorporates other task areas that complement these projects and several that represent accelerated actions leading to earlier site cleanup and closure.

In 1999, under the Integration Project, DOE initiated development of an assessment tool that will enable users to model the movement of contaminants from all waste sites at Hanford through the vadose zone, the groundwater, and the Columbia River and to estimate the impact of contaminants on human health, ecology and the local cultures and economy. This tool is named the System Assessment Capability (SAC).

The approach taken by the SAC is consistent with the methods, characteristics, and controls associated with a composite analysis as described by the Columbia River Comprehensive Impact Assessment (CRCIA) team (DOE-RL 1998). The CRCIA was a study initiated by DOE, the Washington State Department of Ecology, and the U.S. Environmental Protection Agency (EPA) to assess the effects of Hanford-derived materials and contaminants on the Columbia River environment, river-dependent life, and users of river resources. Part I of CRCIA was a study of present-day impacts to the Columbia River from Hanford contaminants. Part II was a suite of requirements for the development of a comprehensive impact assessment for the Columbia River. The two key elements of the SAC approach are 1) ensuring that factors that will dominate the risk are included, and 2) providing an understanding of the uncertainty of the results. Dominant factors were identified through scoping studies and the development of conceptual models for each of the analysis modules used. A stochastic modeling approach was taken to estimate uncertainty in the results. Aspects of uncertainty that could not be included in the calculation were considered in the analysis of the modeling results and discussed in the document presenting initial assessment results (Bryce et al. 2002). The analysis modules included in the SAC parallel those identified by CRCIA and were developed through work group meetings that included regulator and stakeholder participation.

Several key modules were adopted directly from the CRCIA, including the module used to calculate human health impacts (the HUMAN code) and the module used to calculate impacts to ecological species (the ECEM code).

An initial assessment was recently completed with the SAC to demonstrate its functional assessment capability. Future modifications to the tool will be driven by the requirements of specific assessments. Improvements in the results obtained from use of the SAC will be realized as input data are refined through characterization and scientific research. Bryce et al. (2002) reported the results of that assessment, which is the basis for application of the SAC to provide a site-wide perspective of waste disposal and remedial actions in this Hanford Solid Waste Environmental Impact Statement (HSW EIS). Much of the material presented in this appendix has been taken from Bryce et al. (2002).

To simplify the discussion presented in this appendix, the term "SAC" refers to the software package used for this assessment, but it should be noted that the SAC is an evolving and maturing capability.

The initial assessment in fiscal year 2002:

- Modeled the movement of contaminants from 533 locations throughout the Hanford Site representing 890 waste sites through the vadose zone, the groundwater, and the Columbia River.

- Incorporated data on 10 radioactive and chemical contaminants—carbon tetrachloride, cesium-137, chromium, iodine-129, plutonium-239/240, tritium, strontium-90, technetium-99, total uranium (chemical), and uranium (radionuclide).

- Focused on subsurface transport, the Columbia River, and risks to human and ecological health, and the economy and culture.

- Included the geographic region from Rattlesnake Mountain to the Columbia River and from Vernita Bridge to McNary Dam on the Columbia River.

- Included the cleanup actions in Hanford's cleanup plans and agreements as of October 2000.

- Consisted of a stochastic simulation for the period 1944 to 3050 using 25 realizations, thus providing insight into the median response and an initial look at uncertainty.

- Simulated a 1000-year, post-closure period. Three waste forms known to release after that time were not included—immobilized low-activity waste (ILAW), melters, and naval reactor compartments.

For the waste sites located on the Hanford Central Plateau and their associated contaminant plumes, the findings of the initial assessment parallel those of the composite analysis (Kincaid et al. 1998). The results are also consistent with concentrations in environmental media measured by the Hanford Environmental Surveillance Program (Poston et al. 2002). Both the monitoring results and the assessment reported here indicate that Hanford impacts to the Columbia River have peaked and are now declining.

For the purposes of the HSW EIS, the System Assessment Capability (SAC) is a 'best available technology' and, while it remains a tool under development, the SAC Rev. 0 tool is adequate to provide valuable information through quantification of cumulative risks and impacts associated with solid waste disposal at the Hanford Site.

L.1.1 Context of SAC Runs

The principal SAC simulation made in support of the HSW EIS is a series of 25 stochastic simulations run over the period 1944 through 12050 A.D. (that is, a 10,000-year, post-closure period), for the Hanford Site Disposition Baseline (HSDB) scenario. This simulation includes a stochastic representation of inventory, release and transport, and a deterministic representation of exposure and dose. In addition, a median-value input case, based on the median value of each input parameter represented by a distribution in the stochastic model, was simulated.

The HSDB scenario represented in the fiscal year 2002 initial assessment are based on a number of cleanup assumption including waste, debris, and contaminated soil will be removed from the 100 Areas and the remaining soil will meet residential use standards. Similarly, waste, debris, and contaminated soil will be removed from the 300 Areas, but the remaining soil will meet industrial use standards. In this scenario, retrievably stored transuranic (TRU) waste will be recovered, tested to determine waste content, repackaged, and sent offsite for disposal at the Waste Isolation Plant in New Mexico. The waste in Burial Grounds 618-10 and 618-11 will be removed, and the TRU waste will be repackaged and removed from the Hanford Site, while the low-level waste (LLW) will be disposed of in solid waste disposal facilities in the Central Plateau. Ninety-nine percent of the tank waste volume will be recovered from the tanks and a 1 percent residual volume will remain. Losses to the subsurface during waste recovery are assumed to average 30,280 L (8000 gal) per single-shell tank recovered. The recovered tank waste will be separated into low-activity and high-activity fractions. Both waste fractions are assumed to be immobilized. Low-activity waste will be disposed of onsite, while the high-activity fraction will be disposed of in the national repository. All spent fuel also will be stored in a stable configuration for shipment to and disposal in the national repository.

The initial assessment and this analysis assume that, for the duration of the analysis, the future regional and local climate will remain unchanged for the period of the analysis. Furthermore, it is assumed that major engineered structures in the region (for example, the reservoir system on the Columbia River) will remain in place. The recorded climate and environmental response (for example, Columbia River stage and discharge records) since startup of the site operations were used to simulate the period from 1944 to the present. The climate record from 1961 to 1990 was used to represent the future climate. Consequently, the Hanford Site remains a semi-arid, shrub-steppe environment in the simulations. The riparian zone, Columbia River, and river ecosystem are assumed to remain essentially unchanged for the duration of the analysis. Also, the human population will be unchanged and will be based on the current socio-economic setting. Analyses of alternate future climates (for example, global climate change or onset of an ice age and glacial flooding) and potential future events (for example, failure or removal of the reservoir system) are not addressed.

Where the initial assessment addressed the period 1944 through 3050 (that is, essentially a 1000-year, post-closure simulation), simulations for this EIS were carried out over a 10,000-year, post-closure period. Within the SAC, a single transport pathway element, the Columbia River model, is limited to the year 10,000 A.D. in its simulation algorithm, but all other transport pathways (release, vadose zone, groundwater) can execute for the full 10,000-year, post-closure period.

The stochastic simulations supporting the HSW EIS are based on the parameter distributions assembled for the initial assessment. In addition to the environmental pathway and risk/impact model parameters, the inventory and the future disposal and remedial actions assembled for the initial assessment are included. Differences between the inventory used in this extended simulation of the initial assessment and that used in the HSW EIS are described in Section L.2.2.2. Principal differences lie in the methods used to forecast solid waste disposal actions until site closure, both for onsite generators (for example, Waste Treatment Plant contributions) and for offsite generators.

The potential contaminants of greatest concern include technetium-99, iodine-129 and uranium. These contaminants appear in solid waste performance assessments (Wood et al. 1995, Wood 1996) that analyze solid waste disposals in 200 West and 200 East Areas. While the initial application of SAC to the HSW EIS did not include iodine-129, an ability to achieve simulation of iodine-129 is being established. Of necessity, simulation of iodine-129 will include an initial condition for iodine-129 representative of prior releases to the unconfined groundwater, simulation of future releases of iodine-129 per the initial assessment, and superposition of the ILAW contribution to iodine-129 risk and impact. This approach to iodine-129 simulation will include events attributed to past liquid discharges (current groundwater plumes), future solid waste releases, and long-term future releases from immobilized low-activity tank waste. The inventory estimated to exist in the unconfined aquifer, and the estimate of iodine-129 in low-activity tank waste to remain at Hanford will be used in this estimate of the iodine-129 contribution to risk/impact. As in the original 1000-yr initial assessment, simulation of technetium-99 and uranium will use the complete history and forecast of their disposal and begin in 1944 with a clean subsurface environment.

It is unlikely that the plumes from these three classes of release events will superimpose in time. The liquid discharge and unplanned release (e.g., tank leak) sites have created groundwater plumes and will likely continue to release to groundwater during the immediate future. Releases from dry solid waste disposals have some containment (e.g., boxes, drums, plastic bags) and less driving force (e.g., infiltration), and, therefore, they will likely release later than the liquid releases. Finally, the substantially stable and long-term waste forms like vitrified low-activity tank waste will not corrode and release for thousands of years. It is unlikely that peaks from each of these types of release will superimpose in space and time.

L.1.2 Relationship to EIS Calculations

The EIS calculations focus on the impacts associated with alternatives to the disposal of solid waste. The SAC represents a holistic examination of the radioactive and chemical waste legacy of the Hanford Site. For this reason, it can be used to examine the relative risk and impact associated with disposal and remedial action alternatives and the relative role of different segments of Hanford waste—for example, solid waste, past-practice liquid discharges, or tank wastes. Used in this way, the SAC provides an ability

to visualize the change in impact associated with various options and wastes. This kind of cumulative impact assessment provides a larger scale site-wide context from which to view the alternatives and influence disposal decisions.

The EIS calculations provide a detailed evaluation of each specific alternative. The SAC is only able, at this time, to present the single case of an extended analysis (e.g., 10,000 yr post closure) of the HSDB. In essence, the SAC provides an estimate of the contribution made to risk and impact from technetium-99 and uranium from other Hanford waste disposal and remedial actions not explicitly considered in the HSW EIS alternatives, and to contrast that with the contribution from solid wastes.

L.2 Methods and Approach

Historically, DOE has used various tools to assess the effects of waste management and cleanup activities on the environment. Assessments have been performed to address a range of questions. Some assessments have focused on individual waste sites or waste types—for example the assessment performed to evaluate the future performance of the glass waste form proposed for isolating low-activity waste currently in tanks (Mann et al. 2001). Others have looked at contaminants from a variety of sources. The Hanford Environmental Dose Reconstruction Project estimated human health impacts from past releases to the atmosphere and river (Farris et al. 1994) during Hanford operations from 1944 to 1972. The Columbia River Comprehensive Impact Assessment (CRCIA) (DOE-RL 1998) examined ecological and human health effects that might result from the 1990 to 1996 distribution of contaminants in the environment in and near the Columbia River. The composite analysis performed in 1997 considered the impact of selected radionuclides from approximately 280 waste sites in the 200 Areas (Kincaid et al. 1998). In 2001, Bergeron et al. (2001) issued an addendum to the composite analysis that considered additional waste sites on the Central Plateau.

The collective impact of all of the waste that will remain at Hanford, however, had not yet been integrated to provide an understanding of the cumulative effects of Hanford activities on the Central Plateau as well as in the river corridor. The SAC was developed to fill this gap and has benefited from the lessons learned in previous assessments.

The initial assessment and this extension to a 10,000-year, post-closure analysis considers solid waste disposals in the Central Plateau as occurring within aggregated solid waste disposal facilities in the northern and southern portions of the 200 West and East Areas. Annual inventories for each disposal facility within a subregion of the site are aggregated to create an annual solid waste inventory for the subregion. The areal footprints of disposal facilities within a subregion are aggregated to create a total solid waste disposal facility areal footprint. Contaminants from the aggregated disposal facility are released to the unconfined aquifer at the centroid coordinates of the aggregated disposal facility. Thus, use of an aggregated representation of solid waste disposal facilities is an approximation in a number of ways. Notably, the inventory actually placed in individual trenches within each disposal facility is represented as distributed over the entire areal footprint of the disposal facility. Hence, the aggregated inventory is distributed over the aggregated areal footprint of all solid waste disposal facilities in a subregion of the site. Because

of the scale of the aggregation (that is, half an operational area), the centroid of the aggregated area and, hence, the point where contaminants are introduced into the aquifer may lie outside an actual solid waste disposal facility.

The waste form used to represent the disposal of low-activity waste is the vitrified waste form described and analyzed in the Immobilized Low-Activity Waste (ILAW) Performance Assessment (PA) (Mann et al. 2001). The ILAW presents a unit release analysis of the waste inventory, contaminant release, and migration in the vadose zone and groundwater. The contribution of the ILAW source to groundwater and surface water impacts can be estimated by scaling (i.e., for inventory and spatial position). These results can then be superimposed onto the groundwater and surface water impacts predicted for all other Hanford waste sources to achieve a cumulative impact projection. For the initial assessment (Bryce et al. 2002), all contaminants were simulated from 1944 forward in time to estimate the distribution of contamination in the environment. For some contaminants, (e.g., tritium), sufficient process knowledge and data existed to complete a history match against tritium field data. For other contaminants, (e.g., technetium-99, uranium, iodine-129) work is underway to improve our understanding of inventory and mobility to enable improved comparisons to field observations from Hanford's groundwater.

L.2.1 Modular Components of SAC

The SAC development task involved assembling software and gathering the data needed to assess the cumulative impact of radioactive and chemical waste at Hanford. Computer codes that were well tested at the Hanford Site were used when possible, and new software was written when necessary to simulate the features and processes that affect the release of contaminants into the environment, transport of contaminants through the environment, and the impact those contaminants have on living systems, cultures, and the local economy. The components were organized to simulate the transport and fate of contaminants from their presence in Hanford waste sites, through their release to the vadose zone, to their movement in the groundwater, and into the Columbia River. Components such as the groundwater model, the ecological impact component, and the human health component were originally developed and tested for previous Hanford assessments.

The elements of the SAC computational tool include:

- **Inventory Module**—develops an inventory of specific waste disposal and storage locations for the period from 1944 to December 2050 based on disposal records, process knowledge, and the results of tank and field samples. December 2050 was used because it had been identified as the date of site closure. However, for the purposes of this EIS, the Hanford closure date is considered to be 2046. Future analyses will use the current closure date. This module identifies the material scheduled for disposal in offsite repositories including high-activity waste (HLW), TRU waste, and spent fuel.

- **Release Module**—simulates the annual release of contaminants to the vadose zone from the variety of waste types in the modeled waste sites. Waste types explicitly modeled include soil debris wastes as solubility limited desorption, cemented waste as diffusion limited, salt cake tank residuals as nitrate salt dissolution, and graphite cores of production reactors as an empirically defined release.

Because they release after the 1000-year period of analysis, waste types not included in the original SAC design included ILAW, melters, and naval reactor compartments. This module also simulates Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) remedial actions that move waste to the Environmental Restoration Disposal Facility (ERDF) trench.

- **Vadose Zone Module**—simulates the flow and transport of contaminants in the vadose zone, which is the unsaturated sediment between the land surface and the unconfined aquifer. Vadose zone simulations utilize a one-dimensional version of the well-established and documented Subsurface Transport Over Multiple Phases (STOMP) code.

- **Groundwater Module**—simulates the flow of water and the transport of contaminants in the unconfined aquifer that underlies Hanford using the three-dimensional, site-wide groundwater model. Groundwater simulations use the Coupled Fluid, Energy, and Solute Transport (CFEST) code.

- **River Module**—simulates river flow and contaminant/sediment transport in the Hanford Reach from Vernita Bridge downstream to McNary Dam. This model simulates background concentrations and background plus the Hanford Site contribution to enable an assessment of the Hanford Site incremental impact to the Columbia River and its ecosystem. The river model is an extension of the Modular Aquatic Simulation System 2D (MASS2) code developed and applied to support studies of the Snake and Columbia Rivers.

- **Riparian Zone Module**—uses river and groundwater information to simulate the concentration of contaminants in seep or spring water and in the wet soil near the shoreline of the river.

- **Risk/Impact Module**—performs risk/impact analysis in four topical areas: human health, ecological health, economic impact, and cultural impact, with economic and cultural impacts being two new impact metrics for Hanford assessments.

The conceptual illustration of SAC (Figure L.1) portrays a linear flow of information. In general, data flows in the initial assessment in the following manner: the Inventory Module provides input to the Release Module, which provides input to the Vadose Zone, Groundwater, and River Modules. The Vadose Zone Module provides input to the Groundwater Module. Finally, both the Groundwater and River Modules provide input to the Risk/Impact Modules. This version of the SAC conceptual model does not allow feedback among modules and does not include either atmospheric or terrestrial ecological pathways and, hence, receptors.

The data used in the initial assessment came from a variety of sources, including environmental monitoring activities on the Hanford Site, Hanford historical records, a waste site information database, and other geohydrologic and physical property databases. The remediation actions included in the assessment are based on the collection of disposal and remedial actions identified in the Tri-Party Agreement that are planned to occur as the Hanford Site moves toward closure.

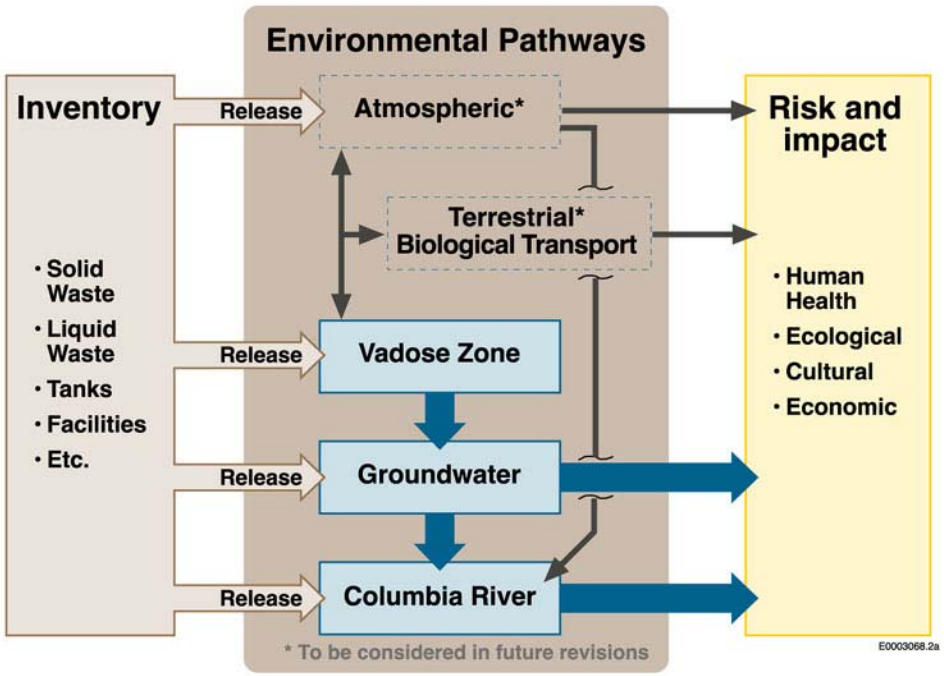

Figure L.1. Conceptual Model of the System Assessment Capability

L.2.2 Inventory

Inventory consists of the quantity of radiological and chemical constituents used and created at the Hanford Site, and their distribution in individual facilities and waste disposal sites. For the initial assessment, inventory was defined as the volume and concentration of contamination introduced annually to waste disposal sites (for example, the solid waste disposal facilities), facilities (for example, the canyon building), and the environment (for example, the vadose zone via liquid discharge sites, the Columbia River via reactor cooling water retention basins). In the initial assessment, export of contaminants to offsite locations was provided by collecting exports at the conclusion of the analysis. The movement of onsite waste from one location to another is included in the Release Module but is limited to the movement of excavated CERCLA wastes to the ERDF trench. Finally, tank waste moves into the Inventory Module of the initial assessment only after it leaks to the environment, is defined as a tank residual, or is recovered from tanks and processed into waste forms that are disposed of onsite or shipped offsite.

The initial assessment included 533 waste site locations throughout the Hanford Site representing 890 waste sites that were identified for consideration. Each of the 890 sites had a likelihood of containing one or more of the contaminants of interest. Some sites were combined, or aggregated, thus reducing the total to 722 sites for analysis. However, of the 722 sites chosen for analysis, only 533 sites were assigned inventories because some waste disposal and unplanned release inventories were further aggregated. For example, individual disposal ditches and ponds were all identified in the list of 722 sites, but the ditch inventories were assigned to the receptor pond. Accordingly, the inventories for the ditches leading to

Gable Mountain Pond, B Pond, and U Pond were assigned zero inventories. The Inventory Module of the SAC generates annual inventories for the selected contaminants at 533 sites for the period from 1944 through 2050, and each of 25 realizations for the stochastic analysis. For the initial assessment, this represented in excess of 782,000 pieces of non-zero inventory data.

Regarding chemicals in solid waste disposals, as in the case of radionuclides it is unlikely that chemical impacts from liquid discharges and solid waste will superimpose in time. It is believed that the majority of chemicals were either discharged to cribs and trenches, or stored in tanks, as opposed to being disposed as solid waste. When the Hanford Site moved away from liquid discharge of chemicals in the late 1960s and early 1970s, substantial chemical waste streams were routed to tanks, (e.g., carbon tetrachloride). Mixed low-level radioactive waste is currently being stored and will be treated prior to disposal under RCRA and past practice CERCLA guidelines to ensure long-term safety. At this time insufficient data and information are available on the chemical inventories in solid waste disposals to perform a site-wide analysis on the scale of the initial assessment. However, it is not clear that chemicals, other than carbon tetrachloride and perhaps chromium, present as substantial a threat to human health as the key radionuclides, technetium-99, iodine-129, and uranium.

L.2.2.1 Initial Assessment Inventory

Methods used to assemble the annual inventory database for all waste sites are described in an appendix to a Composite Analysis addendum issued in September 2001 (Bergeron et al. 2001, Appendix A). Additional detail on the methods used to merge record data and estimates for the Hanford Site inventory were provided by Coony (2002). The addendum to the Composite Analysis includes a summary of the inventory in each waste site at the close of 2000 and at the assumed time of Hanford Site closure in 2050 (Bergeron et al. 2001). The inventory shown in the initial assessment inventory differs from the summary inventory presented in the addendum; however, the data in the addendum provides a representative picture of the Site inventory.

L.2.2.2 Comparison of HSW EIS and Initial Assessment Inventories

The initial assessment inventory was developed over a period of time, beginning in fiscal year 2000 with final entries completed during the spring of 2002. Some of the data entries date from September 1999, the close of fiscal year 1999. The HSW EIS inventory has been developed over a similar time period, but it reflects changes as recent as the summer of 2002. Table L.1 shows a comparison of the initial assessment (SAC) and the EIS as their respective inventories existed in September 2002. The HSW EIS inventories address only wastes assigned to past, present, and future burial grounds, and therefore, while being more current for solid waste, they are not as complete as those assembled for the initial assessment. Table L.1 and the discussion of inventory differences provide a review of the inventories in the two assessments and indicate the relative inventories treated by a soil debris, cement, or liquid release models.

Table L.1. Comparison of Initial Assessment and HSW EIS Inventories

Summary of Technetium-99, Iodine-129 and Uranium Inventories at the Time of Hanford Site Closure							
		Initial Assessment [a]			HSW EIS[b]		
		Tc-99	I129	U	Tc-99	I129	U
Waste Stream	Type	Ci	Ci	Ci	Ci	Ci	Ci
200 East	Solid waste as soil debris	25.3[c]	0.39[c]	0.12	9.1[c]	0.12[c]	32[d,e]
200 East	Solid waste as cement	0.08	0	0	160[d]	0	0
200 East	Tank leaks/residuals	259	0.35	24.8			
200 East	Liquid/UPR	791	0.40	66.2			
200 East	Total Activity	1075	1.14	91.3			
200 West	Solid waste as soil debris	50.2[c]	0.41[c]	209[f]	5.7[c]	.075[c]	150[f]
200 West	Solid waste as cement	1258[c,g]	64.2[c,g]	1837[f,g]	3300[h]	5[h]	1400[f,h]
200 West	Tank leaks/residuals	327	0.61	13.2			
200 West	Liquid/UPR	40.9	0.10	24.7			
200 West	Total Activity	1712	64.9	1803			
ERDF[i] (600-148)		2.6	0.0017	54.0			
SALDS[j] (600-211)	"soil"	0.310	2.17	0.00133			
Graphite Cores (100 Areas)	"core"	0.012	.000089	0			
ILAW (200 East)	"glass"	5929[g]	0[g]	52.97[g]	25,550[k]	22[k]	230[e,k]
Melters (200 East)	"glass"	37.8	0	1.70	38.9	0	1.8
Naval Reactors (200 East)	"rxcomp"	5.18	1.3E-5	0	6		No data
US Ecology (600 Area)	"soil"	60.7	5.45	11390			
Other 200 Area Remaining Onsite[l]		729[m]	0.065[m]	8.6[m]			
Other Areas Remaining Onsite[l]		13.8	0.0044	33.4			

(a) Initial assessment inventory values are median values from a stochastic simulation of the inventory.
(b) lternate A Lower Bound Waste Volume.
(c) The initial assessment includes technetium-99 and iodine-129 inventories estimated using a fuel-ratio method for fission product inventories not reported on original records or prior estimates. The HSW EIS inventories of technetium-99 and iodine-129 include only reported or record values.
(d) The HSW EIS includes inventories of mixed low-level waste (MLLW) that are included elsewhere in the initial assessment inventory for the SAC (see note "m" below).
(e) The HSW EIS includes an inventory of uranium-233 not included in the initial assessment conducted using the SAC.
(f) The initial assessment includes uranium inventories estimated using somewhat different uranium isotopic ratios and estimation methods than used in the HSW EIS.
(g) The initial assessment includes inventory forecasts obtained from a Hanford Tank Waste Operating System (HTWOS) simulation that used potentially out-of-date factors for secondary waste streams.
(h) The HSW EIS includes inventory forecasts obtained from the Solid Waste Information Forecast Tool (SWIFT) that includes a life-cycle forecast of the composition of secondary waste streams from tank waste.
(i) Environmental Restoration Disposal Facility (ERDF).
(j) State Approved Land Disposal Site (SALDS).
(k) The HSW EIS includes inventory forecasts obtained from the ILAW performance assessment (PA) (Mann et al. 2001) for isotopes, and from a current estimate of technetium-99 that will be routed to low-activity waste disposal.
(l) Does not include waste listed above.
(m) The initial assessment includes inventories of MLLW at the Hanford Site that will be routed though the Radioactive Mixed Waste Storage Facility prior to disposal onsite.

The SAC was applied in the HSW EIS to generate both a stochastic simulation and a median-inputs deterministic simulation. The inventory values reported for the initial assessment in Table L.1 are median values of the stochastic distribution. Thus, a varied inventory is analyzed, and each of the 25 realizations is based on a Latin hypercube selection procedure. For sites not modeled using process knowledge and a stochastic simulator (Simpson et al. 2001), site-specific inventories prior to 1970 are modeled as twenty-fold uncertain; that is, the maximum is approximately 20 times the inventory database value, and the minimum is approximately one-twentieth of the inventory database value. After 1970, the inventories for these sites are modeled as twofold uncertain; that is, the maximum is approximately twice the inventory database value, and the minimum is approximately half the database value.

The inventory analyzed by the site-wide groundwater model and the unit release approach in the HSW EIS was provided by Fluor Hanford. The inventory analyzed using the SAC tool is based on available records and was augmented with estimated inventories for fission products (for example, technetium-99 and iodine-129) and uranium isotopes where they are absent from the record. The augmented values are only estimates and should not be considered record values.

There are differences in the compilations shown in Table L.1. Solid waste deposits in the 200 East and 200 West Areas differ primarily as follows: 1) the initial-assessment technetium-99 and iodine-129 inventories include fuel-ratio estimates of this fission product, 2) the initial-assessment uranium inventories include estimates based on uranium-isotopic ratio methods of estimation that differ from those of the EIS, 3) the HSW EIS uranium inventories include MLLW inventories that are accounted for elsewhere in the initial assessment, and 4) HSW EIS solid waste disposal facility uranium inventories include uranium-233, which was omitted from the initial assessment.

A major difference in inventories in 200 West Area solid waste disposal facility deposits and in ILAW and melter deposits lies in the use of different resources to estimate future disposals and secondary wastes from the processing and solidification of high- and low-activity wastes at Hanford. The initial assessment relied on the Hanford Tank Waste Operation System (HTWOS) model that relied on a suite of potentially out-of-date factors to estimate secondary waste stream composition. The HSW EIS relied on current ILAW and melter inventories. Inventories with the greatest differences are either simulated as cement waste forms that release relatively slowly, (for example, 200 West Area solid waste cement), or are not simulated by the initial assessment, (for example, ILAW and melter waste). A difference of approximately 2000 Ci in technetium-99 exists between the two estimates of secondary technetium-99 wastes. Similarly, a difference of approximately 60 Ci in iodine-129 exists. These differences will be reconciled as projections are updated; however, all of this waste would be disposed of in cement to minimize the hazard. In the analyses undertaken for both the initial assessment and the HSW EIS, the majority of the future uranium inventory is disposed of in cement to minimize the hazard. Finally, because of the original design objectives of the SAC (that is, a 1000-year analysis), the initial assessment does not include the glass release model(s) necessary to forecast the long-term release of the ILAW and melter wastes. Hence, the influence of ILAW and melter inventories is not included in the initial assessment results, or in the extended, that is 10,000-year, initial assessment presented here. Naval reactor compartments are also omitted from SAC analyses at this time. However, for the greatest of these inventories, ILAW, their influence is introduced to the cumulative assessment by superimposing the results of the ILAW PA (Mann et al. 2001) onto the initial assessment result.

In addition to the values indicated in Table L.1 the SAC simulation had 7.15 Ci of iodine-129 in spent fuel and includes an estimated 18.9 Ci of iodine-129 released to the atmosphere during the operation of chemical separation plants. The 64 Ci of iodine-129 in 200 West "solid waste as cement," is almost entirely from HTWOS analysis byproduct streams from vitrification (that is, spent resins, and ILAW and HLW waste streams (not glass)). At the time the initial assessment inventory was assembled, the HTWOS processing fractions had no iodine going to any immobilized waste product (that is, ILAW, melters, or HLW). The median value iodine-129 inventory for the initial assessment had a total of about 103 Ci.

Inventories included in the initial assessment for the commercial low-level radioactive waste disposal site operated by US Ecology at Hanford are based in part on the published State of Washington SEPA Draft EIS (Washington State Department of Health and Washington State Department of Ecology 2000) and the Closure Plan for the site published by US Ecology, Inc. (1996). The State of Washington is now reviewing the inventory for the commercial site during its early years of operation. Hanford staff are in contact with a representative of the State Department of Health, and as soon as an updated inventory is available it will be incorporated into Hanford assessments. Certainly, uranium inventories for the commercial low-level radioactive waste disposal site appear to be relatively high in the initial assessment.

L.2.3 Release

Release is the rate at which radioactive and chemical contaminants find their way into the environment. The SAC Release Module handles liquid releases and releases from solid waste forms. It is important to note that because the initial assessment was originally designed as a 1000-year analysis, several waste forms that will not be released in this period were not analyzed and were not analyzed in this extended 10,000-year, post-closure analysis even though they may be released in the 10,000-year time frame. These waste forms include naval reactor compartments, immobilized low-activity waste, and components of melter systems. Liquid discharges, liquid unplanned releases including tank leaks, and future tank losses are handled as a simple pass-through to the vadose zone or the Columbia River. The solid waste forms are primarily in solid waste disposal facilities including past-practice sites (pre-1988), active sites (post-1988), and the ERDF. Other solid waste includes residual waste in the single-shell tanks, the graphite cores of the retired production reactors, and concrete and cement waste forms associated with caissons, canyon buildings, and grouted waste.

The Release Module applies release models to waste inventory from the Inventory Module and also accounts for site remediation activities (for example, waste movement) as a function of time. The resulting releases to the vadose zone, expressed as time profiles of annual rates, become source terms for the Vadose Zone Module. Radioactive decay is accounted for in all inputs and outputs of the Release Module. The Release Module is implemented as the VADose zone Environmental Release (VADER) computer code.

L.2.3.1 Conceptual Model

Waste containment facilities have a number of features that influence the rate at which contaminants can be released from waste. The waste may be placed in a trench or may reside in a tank. The trench,

tank, or other engineered structure may have features that serve as barriers to prevent infiltrating water from making contact with and transporting contaminants from the waste to the vadose zone. Waste inside an engineered structure (for example, a trench) may also be contained in a waste package (for example, a metal drum or high-integrity concrete container). The drum or concrete container acts as an additional barrier that prevents transport of the contaminants from the waste. Major containment materials for Hanford waste are concrete, steel, and bituminous layers and coatings. The stability and permeability of concrete materials change over time, and likewise, time affects the features that dominate water or contaminant migration in containment materials. Surface covers on an engineered system and liners (geomembrane and geosynthetic) and leachate collection systems at the bottom of a system further restrict infiltrating water from transporting contaminants to the vadose zone. Surface covers are particularly important because migration of infiltrating pore water may be limited as long as the cover maintains its integrity. Individual waste sites have one or more of these features. However, none of the waste sites in the initial assessment had all of the features in the conceptual model.

A number of key processes govern how much contaminant at any given time is released from the waste to the infiltrating water. One process is the affinity of contaminants to be retained by the waste (for example, sorption to soil or waste material). Another process is the ability of waste to dissolve and, in some cases, to form new precipitates, thus allowing some contaminants to be released to the infiltrating water while others remain trapped in the precipitated solids. Release from the waste may also be limited by the solubility of the contaminant in the infiltrating water.

Water infiltrating an engineered system may contact and react with fill materials (for example, soil, basalt, or grout), containment materials in various states of degradation, and different types of waste. Reaction with these materials will change the water chemistry and the physical and hydraulic properties over time. The water composition, pH, and redox state at any given time will influence the extent to which these processes influence contaminant release from the waste.

L.2.3.2 Implementation Model

The Release Module accounts for releases that occurred in the early years of Hanford operations, releases that may be expected while the Site is being cleaned up over the next several decades, and future releases that may continue until the entire inventory is released. The Release Module relies on several sources for input. Input from the Inventory Module includes contaminant mass (for chemicals) and activity (for radionuclides) deposits. Some of the release models (that is, soil-debris, cement) require site or waste feature information (that is, site cross-sectional area, site volume, or waste surface area or volume). Recharge rate is an important parameter for the salt cake and soil-debris models. Key process parameters are distribution coefficient (soil-debris model), solubility (soil-debris, C_{sol}, and salt cake models), diffusion coefficient (cement model), and fractional release rate (reactor block model).

To capture uncertainty in the SAC simulations, contaminant inventories and numerical model parameters are expressed in terms of statistical distributions. Each realization of the initial assessment used sample parameter values for randomly distributed variables such as bulk soil density, soil moisture content, sorption or distribution coefficient, salt cake density, and cement diffusion coefficient. Other model parameters were held to constant values over all realizations.

L.2.3.3 Numerical Models Relevant to HSW EIS

Soil-Debris Model

The soil-debris model is used to model contaminant release from unconsolidated wastes mixed with soil. Source zones composed of this waste-form type are permeable to percolating water; thus, all surfaces of the waste come in contact with the percolating water as it passes through the zone in a manner similar to the way infiltrating water passes through natural vadose zone material. The soil-debris model is applied to the release of contaminants from all solid waste disposal facilities, including the ERDF, and the commercial low-level radioactive waste disposal facility operated by US Ecology, Inc.

For the SAC initial assessment, the model used the high-impact values of the distribution coefficient parameter (K_d) associated with the vadose zone nearest the disposal facility. For solid waste disposal facilities, the K_d category used by the soil-debris model is that associated with sites that are low organic, low salts, and near neutral pH. The K_d best-estimate values for this category were 0 mL/g, 0.5 mL/g, and 3 mL/g for technetium-99, iodine-129, and uranium, respectively.

For radionuclides for which no specific solubility values were available, the aqueous solubility was fixed at an arbitrarily high default value ($1x10^{10}$ mg/L) so that the soil-debris model automatically selects algorithms for sorption (K_d) control in these cases (Kincaid et al. 1998). Technetium-99 solubility ($1x10^{10}$ mg/L or $1.7x10^{2}$ Ci/cm^3) was assigned using this approach. Iodine-129 solubility ($1x10^{10}$ mg/L or 1.77×10^{0} Ci/cm^3) was also assigned using this approach. Uranium solubility (86.9 mg/L or $2.95x10^{-11}$ Ci/cm^3) was estimated in Hanford groundwater assuming that the solid controlling uranium solubility was $UO_2 (OH)_2 \bullet H_2O$ (Wood et al. 1995).

In the simulation runs, K_d, θ_w, and ß were treated as stochastic over the 25 realizations, and Q_w and C_{sol} were fixed to a constant value for all analytes except tritium. For tritium, K_d was set to zero over all realizations.

Sites with soil wastes include the '118,' '218,' and '618' sites listed in Bergeron et al (2001).

Analytical Solution for Instantaneous Release—Soil-Debris Model

The rate of loss of contaminant for a given contaminant by the soil-debris model is given by Kincaid et al. (1998) as:

$$dM / dt = - Q_w A C_w$$

where $C_w = C_{sol}$ when the release process is solubility controlled and $C_w = M /(\theta RAh)$ when the release process is desorption-controlled where:

$$R = 1 + (\beta K_d) / \theta$$

Switching régimes is controlled by comparing the remaining mass with the maximum mass M_{max} consistent with an aqueous phase saturated with the contaminant. If M, the mass remaining in the waste form, is larger than the quantity M_{max} where:

$$M_{max} = \theta R C_{sol} A h$$

the release process is considered to be solubility controlled. Otherwise, it is considered desorption controlled.

Coupling the soil-debris model with an aggregated waste site representation leads to a lower calculated waste concentration, a reduced likelihood of a solubility-controlled release, and a greater likelihood of a desorption-controlled release. Because the release occurs over a larger area than really occupied by the waste deposit, the calculated release is a function of a greater amount of infiltrating water contacting the waste. Thus, all contaminants are leached and for mobile contaminants such as technetium-99 that are not solubility controlled, the release is greater for an aggregated site approach.

Definitions

- M_{max} is the maximum amount of contaminant possible in the source zone (in Ci or kg) without a precipitated phase.
- M = M(t) is the current quantity of contaminant contained in the source zone (Ci or kg).
- Q_w is the recharge rate for the site in cm/yr. Q_w can be considered to be constant, or it can be time-dependent based on site climate and remediation activities.
- hA is the surface area of the soil waste form exposed to the release mechanism (cm^2).
- h is the depth of the waste form in the site (cm).
- C_w is a coefficient expressing the effective release of the contaminant (Ci/cm^3 or kg/cm^3).
- C_{sol} expresses aqueous solubility of the contaminant in Ci/cm^3 or kg/cm^3.
- R is either a retardation factor or a soil apportionment factor (unitless) that depends on the following factors:
  - ß Soil bulk density in g/cm^3
  - K_d Sorption factor (cm^3/g)
  - θ Soil volumetric content of water in soil (unitless fraction).

- dM/dt is the rate of loss of contaminant from the source zone (the rate contaminant crosses the soil waste form boundary and enters the environment).

- t is the elapsed time (years) from the beginning of release from containment.

C_{sol} (Solubility) Model

The C_{sol} model is the independently operated solubility-controlled analytical solution component of the soil-debris model. As such, it is applied to the same types of solid wastes that are applied to the soil-debris model. The difference is that the process represented by the C_{sol} model is that of a constant concentration release. The concentration at which a contaminant is released from a waste often is at its solubility limit in some aqueous medium (for example, groundwater or grout leachate) but is not a requirement. This is different from application of the same analytical solution within the soil-debris model in which the model determines the process (solubility controlled vs. sorption controlled) that is the appropriate for application at any time within a simulation. In addition, release is always at what is considered to be the solubility limit of the contaminant in the aqueous media of interest. The analytical solution and key parameters are the same as those described in the previous section for the solubility-controlled analytical solution component of the soil-debris model.

Initial application of this release model within the SAC Release Module was undertaken to provide a comparative evaluation of uranium release from a cemented waste form using three different release models (see Section L.2.3.4).

Assume that a solubility-controlled release was prescribed for several scales of disposal from aggregated areas to individual waste trenches, and that each disposal scale contained the same inventory. The larger the waste site area, the greater the infiltrating water quantity contacting waste, and the greater the mass or curie flux from the waste site and the more rapid the release.

Cement Model

The cement model is generally applied to cementitious waste forms. Knowledge of the total external surface area and the volume of the waste form is required. The ratio of area-to-volume is assumed to be constant—that is, the waste form is assumed not to degrade in terms of shape over the duration of the contaminant release process. In the SAC initial assessment, the cement model was used to simulate release of contaminants from cementitious wastes within selected solid waste disposal facilities. Delay of contaminant release from containerized waste can be accomplished with the current capability by arbitrarily assigning a time of delay. In the SAC initial assessment, however, no credit was taken for container integrity. Plans call for incorporating one or more models into a future revision of the SAC capability that will accommodate delay of release from contained waste based on specific processes (for example, corrosion of metal).

The range in diffusion coefficient values ($1.58x10^{-4}$ cm/y to $1.89x10^{-3}$ cm/y) used in the SAC initial assessment for technetium-99 was obtained from recent laboratory work (Mattigod et al. 2000). The diffusion coefficient for uranium ($3.15x10^{-5}$ cm/y) was obtained from Serne et al. (1992). In the simulations, the diffusion coefficient for technetium-99 was stochastic; for uranium, it was set to a constant for all realizations.

Sites containing cementitious wastes include the '202,' '221,' 224,' and '276' sites listed in Bergeron et al. (2001).

Analytical Solution for Instantaneous Release—Cement Model

The contaminant release mechanism of the cement model is diffusion in the pore water of the solidified waste material to the outer surface of the waste form. The rate of loss of contaminant for a given contaminant is given by Kincaid et al. (1998) as:

$$dM/dt = Mo(A/V)\sqrt{D/_{\pi}t}$$

where:
- M_0 = the original quantity of the contaminant contained in the cement (Ci or kg)
- M = current quantity of the contaminant contained in the cement (Ci or kg)
- A = the surface area of the cement structure (cm^2)
- V = the volume of the cement structure (cm^3)
- D = the diffusion coefficient of the contaminant (cm^2/yr)
- t = the elapsed time (years) from the beginning of release from containment
- dM/dt = the rate of loss of contaminant from the cement waste form
- π = 3.14159.

Note, the original quantity M_0 can be seen as a function of concentration (kg/cm^3 or Ci/cm^3) and volume (cm^3).

With regard to the scale of the disposal, assuming the aggregated area of an aggregated volume is simply the exterior surface of the volume, the larger the disposal area – the smaller the ratio of area to volume (A/V) in the equation above. Accordingly, if the contaminant mass or curies and the diffusion coefficient are unchanged for multiple scales of waste site, then the larger aggregated site will exhibit a lower release rate.

Containment

The release models implemented in the current version of SAC have no provisions for specifically modeling containment of wastes, such as high-integrity steel containers. The models do have provision for delaying release to a specific start year (that is, the STARTREL argument in the MODELS keyword). The default start year is the year the waste begins to be deposited at the Site. In the initial assessment, STARTREL was set to 1944 throughout the simulation, so for the initial assessment, the release mechanism was active as soon as wastes were deposited.

L.2.3.4 Comparison of Release Model Parameters

A comparison of key source-term release models (that is, soil debris, solubility-controlled, and cement) and values of key parameters used in the SAC analysis, the HSW-EIS analysis (described in Appendix G), and the Solid Waste Burial Ground Performance Assessments (SWBG-PAs) for the 200 West and East Areas (as described by Wood et al. [1995, 1996]) is summarized in Table L.2. The three constituents addressed are technetium-99, iodine-129, and uranium. This summary of parameter values coupled with the release model formulations of the preceding section allow a comparison of relative release characteristics included in the three assessments. The parameter values shown here are somewhat generic and not necessarily related to specific waste streams, and, therefore, could be changed according to specific waste disposal conditions for application in specific wastes and especially for regulatory compliance simulations (that is, a performance assessment for a specific disposal).

There are several key differences in the way these different analysis approaches address selective contaminant releases from the source term. The SAC analysis differs from the other two analyses in the way that uranium is released from LLW. For non-cemented waste, the SAC analysis uses a soil debris model coupled with uranium specific solubility-limits to simulate uranium release. For cemented wastes, the SAC analysis uses a cement (that is, diffusion-controlled) release model to simulate uranium release. In contrast, the release of uranium in HSW-EIS analysis and the SWBG PAs both rely on a solubility-controlled release model with uranium specific solubility limits depending on whether the uranium inventory is contained in non-cemented wastes or in cemented wastes (for example, 64 mg/l for non-cemented wastes and 0.23 mg/l for cemented wastes).

The SAC application of the cement model to technetium-99, iodine-129, and uranium releases assumed a cemented waste and a surface area to volume ratio based on a waste volume that constituted a number of aggregated burial ground sites. In contrast, the HSW-EIS and SWBG-PA analyses rely on a conceptualization of surface area-to-volume (A/V) ratio based on the surface area and volume of individual waste containers (for example, individual steel barrels, boxes, high integrity containers that would contain grouted wastes). As a result, the surface A/V ratio for the SAC source term was up to 10 times lower than those reported for HSW-EIS and SWBG-PA analyses. A lower release of technetium-99, iodine-129, and uranium from the SAC analysis would be expected based on this difference alone. However, when the diffusion coefficient is roughly an order of magnitude higher in the SAC application, the lower A/V ratio is partially offset by the higher diffusion coefficient.

From the formulations of the soil debris model, which is the release model associated with early solid waste disposals at Hanford (that is, pre-1970 wastes), it is apparent that the use of larger aggregated areas as opposed to burial ground, trench, or caisson scales to represent waste, leads to lower initial concentrations of waste but exposes waste to greater infiltration, and, hence leaching. Use of aggregated representations and the soil debris model tends to release waste more rapidly than would occur if simulations were conducted on the burial ground or trench scale.

Table L.2. Comparison of Selected Values of Key Parameters Used in Source Term Release Models for the System Assessment Capability Analysis Described in this Appendix, the HSW EIS Analysis Described in Appendix G, and the Solid Waste Burial Ground Performance Assessments for 200 West and 200 East Areas Described in Wood et al. (1995, 1996).

	System Assessment	HSW EIS	Solid Waste Performance Assessment
Source-Term Release Models			
Soil-Debris Model			
Model or Zone/Parameter	**Data/Statistical Treatment**		
Volumetric Moisture Content (%)	0.0594 ± 0.0310 [(a)] (mean/standard deviation, normal distribution)	0.05	0.05
Bulk Density (g/cm^3)	1.535 ± 0.1085 [(a)] (mean/standard deviation, normal distribution)	1.6	1.5
Waste Thickness (m)	5.349 [(b)] (deterministic)	6	4.5
K_d uranium (mL/g)	Low organic/low salt/near neutral, high impact: (best estimate, min and max) best estimate: 3, min: 0.1, max: 500	Mobility Class (K_d=0.6) covering constituents with K_ds between 0.6 and 0.9999	Mobility Class (K_d=0.0) covering constituents with K_ds between 0.0 and 0.9999
K_d technetium-99 (mL/g)	Low organic/low salt/near neutral, high impact: (best estimate, min and max) best estimate: 0; min: 0; max: 0.1	Mobility Class (K_d=0.0) covering constituents with K_ds between 0.0 and 0.5999	Mobility Class (K_d=0.0) covering constituents with K_ds between 0.0 and 0.9999
K_d iodine-129 (mL/g)	Low organic/low salt/near neutral, high impact: (triangular distribution, mode, min and max) median: 0.5; min: 0; max: 15	Mobility Class (K_d=0.0) covering constituents with K_ds between 0.0 and 0.5999	Mobility Class (K_d=0.0) covering constituents with K_ds between 0.0 and 0.9999
Solubility; uranium (mg/L)	86.9 (2.95 x 10^{-11} Ci/cm^3) [(d)] (deterministic) (non cemented wastes)	none	none
Solubility; technetium-99 (mg/L)	1 x 10^{10} (1.7 x 10^2 Ci/cm^3) [(e)] (deterministic) (non-cemented wastes)	none	none
Solubility; iodine-129 (mg/L)	1 x 10^{10} (1.77 x 10^0 Ci/cm^3) [(e)] deterministic (non-cemented wastes)	none	none
Solubility-Control Model			
Model or Zone/Parameter	**Data/Statistical Treatment**		
Solubility; uranium (mg/L)	86.9 (2.95 x 10^{-11} Ci/cm^3) [(d)] (deterministic) (non cemented wastes)	64 (non-cemented wastes); 0.23 (cemented wastes)	64 (non-cemented wastes); 0.23 (cemented wastes)
Solubility; technetium-99 (mg/L)	1 x 10^{10} (1.7 x 10^2 Ci/cm^3) [(e)] (deterministic) (non cemented wastes)	none	none
Solubility; iodine-129 (mg/L)	1 x 10^{10} (1.77 x 10^0 Ci/cm^3) [(e)] (deterministic) (non cemented wastes)	none	none
Cement Model			
Model or Zone/Parameter	**Statistical Treatment**		
Area to Volume Ratio (m^2/m^3)	0.378 [(k)]	1.55 to 1.93	5.33 [(i)]
Diffusion Coefficient; uranium (cm^2/y)	3.15 x 10^{-5} (1 x 10^{-12} cm^2/s) [(e,f)] (deterministic)	NA	NA
Diffusion Coefficient; technetium-99 (cm^2/y)	(uniform distribution, median, min, max) median: 1.02 x 10^{-3}, min: 1.58 x 10^{-4}, max: 1.89 x 10^{-3} [(g)]	3.15 x 10^{-4}	3.15 x 10^{-5} to 31.5 [(j)]
Diffusion Coefficient (iodine 129) (cm^2/y)	3.5 x 10^{-5} [(g)]	3.15 x 10^{-5}	3.15 x 10^{-5} to 31.5 [(j)]

(a) Values based on statistical treatment of individual data points measured or calculated over a depth ranging from 0- to 20-ft values calculated from bulk density and moisture content data from Fayer et al. (1999).
(b) An average height calculated for burial ground sites based on available height information in the WIDS database.
(c) Based on revision of K_ds in Kincaid et al. (1998) resulting from a recent compilation and evaluation of distribution coefficient data in Hanford sediments (Cantrell et al. 2002).
(d) Estimated solubility in Hanford groundwater assuming solid controlling solubility was $UO_2(OH)_2 \cdot H_2O$ (Wood et al. 1995).
(e) Default value from Table D.2 of Kincaid et al. (1998).
(f) Recommended value (default) for generic grout performance assessment when actual grout-specific data is lacking (Table 6, Serne et al. 1992).
(g) Based on results obtained from Mattigod et al. (2000).
(h) Best estimate K_d values after Cantrell et al. (2002).
(i) Values as low as 1.7 m^2/m^3 have been used in subsequent waste stream specific analyses.
(j) A range of values was considered for an unspecified constituent in the PA analysis (Wood et al. 1995).
(k) Based on all cemented waste placed in aggregate area 218-W@T-6-12 (SAC rev. 0).

L.2.4 Vadose Zone Module

The Vadose Zone Module is designed to simulate the transport and fate of contaminants as they move through the hydrogeologic region that extends from the land surface to the regional water table. Kincaid et al. (2000) identified the STOMP computer code (White and Oostrom 1996) as the code for the Vadose Zone Flow and Transport Module for SAC. Inputs to the Vadose Zone Module come primarily from the inventory and release elements, including recharge, and the mass flux and concentrations of the selected constituents. Other inputs include the effectiveness and timing of remedial actions that might either reduce the mass and/or concentration of contaminants in the vadose zone or that might reduce the flux of deep infiltrating moisture (that is, capping). These inputs include infiltration rates from both natural events (for example, precipitation) and operational activities (for example, excavation or capping). A few major hydro-stratigraphic units that are of uniform thickness and horizontal with homogeneous and isotropic properties were used to represent each site. Hydraulic and geochemical parameters for each hydro-stratigraphic unit are represented by stochastic distributions that reflect the uncertainty in measured properties. Definitions of the hydro-stratigraphy and the associated hydraulic, transport, and geochemical properties of the one-dimensional soil column were based on existing geologic, soil physics, and geochemical databases.

L.2.4.1 Distribution Coefficients (K_ds) for Technetium-99 and Uranium

The SAC initial assessments used K_d values that were assigned to each hydrogeologic unit in a manner similar to that done for the Composite Analysis (Kincaid et. al. 1998). The waste characteristics were assumed to dominate the near-field mobility of the contaminants in the vadose zone. After being in contact with vadose zone sediments and soil water for some distance, the waste undergoes a change in its mobility based on buffering of the contaminant solution by the vadose zone sediments. Thus, distribution coefficients were defined separately for each contaminant in the upper vadose zone (near-field or high-impact zone) and in the lower vadose zone (far-field or intermediate-impact zone) (Kincaid et. al. 1998).

Distribution coefficient zones were defined as either high-impact or intermediate-impact depending on the nature of the contaminant. Zones in which the organic concentration, pH, or salt concentration in the fluids may have affected the K_d values were designated high-impact. Zones in which the acidic or basic nature of the wastes was estimated to have been neutralized by the natural soil were designated intermediate-impact. Kincaid et al. (1998) estimated the depths of this transition zone by examining the peak location of beta/gamma contamination, as presented by Fecht et al. (1977), for 200 Area cribs receiving very acid or high-salt/very basic waste. In general, these transition depths ranged from 10 to 40 m (33 to 130 ft). Given the limited data available on which to base further interpretations on the depths of transition and the desire to simplify the numerical simulations, a slightly different approach was used here. Generally, the hydrogeologic unit into which waste streams were introduced was designated as high-impact regardless of waste stream characteristics. If those hydrogeologic units were thin (for example, less than 10 m), then the hydrogeologic unit immediately below that into which the waste stream was introduced was also designated as high-impact. All other hydrogeologic units lower in the profile were designated intermediate-impact. This approach kept the numerical simulations relatively simple by using the existing number of hydrogeologic units (that is, new layers did not need to be added to make the K_d change where it might have occurred within a single hydrogeologic unit). At the same time, the depths of

change, corresponding to the thickness of the hydrogeologic units, are still on the same scale (10s of meters) as those used by Kincaid et al. (1998). A summary of the K_d values used for technetium-99 and uranium is presented in Tables L.3 and L.4, respectively.

L.2.4.2 Vadose Zone Strata and Hydraulic Properties

Of the more than 2600 waste sites at Hanford cataloged in Waste Information Database System (WIDS), a subset of 533 was selected for simulation in the initial assessment. Because of the aggregation of solid waste disposal facilities, unplanned releases, and various liquid discharge sites into fewer global waste sites within operational areas or portions of operational areas, these 533 sites represent 890 waste sites.

Geologic Profiles

Each of these sites were assigned to one of 64 base templates defined on the basis of 1) the type of waste site, 2) its geographic location (that is, area/geology), and 3) the characteristics of the waste stream.

Table L.3. Technetium-99 K_ds in ml/g

Waste Chemistry	Vadose Zone			Groundwater	Reparian Zone
	Near-field (High Impact)	Far-Field (Intermediate Impact)			
		Sand	Gravel		
All	**0** (0-0.1)	**0** (0- 0.1)	**0** (0-0.01)	**0** (0-0.1)	**0** (0-0.0001)
Values are listed as **best** (minimum–maximum).					

Table L.4. Uranium K_ds in ml/g

Waste Chemistry	Vadose Zone			Groundwater	Reparian Zone
	Near-field (High Impact)	Far-Field (Intermediate Impact)			
		Sand	Gravel		
High Organic/Very Acidic; Chelates/High Salts; Low Organic/Low Salts/Acidic	**0.2** (0-4)	**0.8** (0.2-4)	**0.08** (0.02-0.4)	**0.8** (0.2-4)	**0.0008** (0.0002-0.0004)
High Organic/Near Neutral; Very High Salt/Very Basic; Low Organic/Low Salt/ Near Neutral	**0.8** (0.2-4)				
Values are listed as **best** (minimum–maximum).					

Generalized hydro-stratigraphic columns were specified for each of the 13 geographic areas. These columns were assembled from existing information including:

- drillers' logs, geologists' logs, and geophysical logs
- published interpretive depths to the top and bottom surfaces of hydrogeologic units
- surface elevations (to convert hydrogeologic unit depths to elevations)
- elevation of the 1944 water table (to define the bottom of the vadose zone prior to waste disposal).

The generalized hydrostratigraphic units used in this study are summarized in Table L.5.

Table L.5. Summary of Hydrogeologic Units Used in This Study

Hydrogeologic Units	Facies/Subunit	Description
Not Applicable	Backfill	Poorly sorted gravel, sand, and silt derived from the Hanford formation and/or Holocene deposits
Holocene	Eolian	Dune sand and silt
Hanford formation	Silt-dominated	Interbedded silt and fine to coarse sand
	Fine sand-dominated	Stratified fine sand with minor pebbles and minor laterally discontinuous silt interbeds
	Coarse sand-dominated	Stratified coarse sand with minor pebbles and minor laterally discontinuous silt interbeds
	Gravelly sand	Cross bedded, interstratified coarse sand with up to 30 wt% very fine pebble to cobble
	Gravel-dominated	Cross bedded, interstratified coarse sand and gravel with greater than 30 wt% very fine pebble to boulder
	Undifferentiated	Undifferentiated sand and gravel with minor discontinuous silt interbeds.
Plio-Pleistocene Unit	Silt/sand dominated	Very fine sand to clayey silt sequence. Interstratified silt to silty very fine sand and clay deposits
	Carbonate rich	Carbonate-rich sequence. Weathered and naturally altered sandy silt to sandy gravel, moderately to strongly cemented with secondary pedogenic calcium carbonate.
Ringold Formation	Fluvial sand (member of Taylor Flat)	Interstratified sand and silt deposits
	Fluvial gravel (member of Wooded Island, subunit E)	Moderate to strongly cemented well-rounded gravel and sand deposits, and interstratified finer-grained deposits.
	Overbank/Lacustrine deposits (lower mud sequence)	Predominantly mud (silt and clay) with well-developed argillic to calcic paleosols.

In general, the depth and thickness of each hydrogeologic layer (strata) for each geographic area were taken from published maps and cross sections. The estimated average strata thickness was used for the generalized columns extending from the surface to the 1944 water table (Kipp and Mudd 1974). Because the sum of the average thickness did not always equal the distance from the land surface to the groundwater, small adjustments were made to the average strata thickness.

Hydraulic Properties

Hydraulic property data were primarily taken from Khaleel and Freeman (1995) as supplemented by Khaleel (1999) and Khaleel et al. (2000). Because this data set is rather limited in regards to the spatial location of samples and the soil types represented, individual stochastic data sets were selected to represent each hydrogeologic strata present in the 13 geographical areas. Care was taken to ensure that the soil classifications for which hydraulic property data was available could be correlated to the sediment facies within each template.

The statistical distributions of van Genuchten model (van Genuchten 1980) parameters, saturated hydraulic conductivity, and bulk density data were taken primarily from Khaleel and Freeman (1995) and Khaleel et al. (2000), and the distributions for longitudinal dispersivity were primarily taken from Ho et al. (1999). Values for residual saturation (S_r) were calculated by dividing the raw residual water content (θ_R) by the raw saturated content (θ_s), as provided by Khaleel and Freeman (1995). Effective porosity is assumed to be equal to the saturated water content (θ_s). Note that all model nodes within a single hydrogeologic unit are assigned the same hydraulic properties for a single realization.

L.2.4.3 Surface Covers

The SAC incorporates recharge estimates into the STOMP model to provide deterministic values that change stepwise as the surface cover changes and to represent the degradation of engineered covers following their design life. The recharge rates (actually, deep drainage rates) used for the SAC were estimated for all surface conditions under consideration for the initial assessments. These conditions included four different barrier designs, degraded barriers, the natural conditions that surround the barriers, and the unique conditions created by human activities (for example, facility construction, gravel-covered tank farms). Recharge estimates were based on the best available data (Fayer and Walters 1995, Fayer et al. 1999, Murphy et al. 1996, Prych 1998).

Barrier Recharge Estimates

Recharge through engineered surface covers was estimated based on the Focused Feasibility Study (FFS) conducted by DOE-RL (DOE-RL 1996). The FFS was conducted to determine the barrier needs at Hanford and to identify a set of barrier designs to meet those needs. Table L.8 identifies the four barrier designs that were proposed. According to the FFS, the modified Resource Conservation and Recovery Act (RCRA) Subtitle C design will be the predominant barrier type. DOE-RL (1996) used the Hydrologic Evaluation of Landfill Performance (HELP) model to simulate the recharge rate through the Hanford Barrier, modified RCRA barriers, and the standard RCRA barriers. The estimates ranged from 0.2 to 0.8 mm/yr., assuming that the annual mean precipitation remained at 160 mm/yr (6.3 in./yr).

Subsequent to the FFS, additional data and model results became available. As a result, the recharge rates for the barriers were updated and are reflected in Table L.6.

No guidance is available for specifying barrier performance after the design life. However, an immediate decrease in performance is not expected, and it is likely that some of these barriers will perform as designed far beyond their design life. Without data to understand and predict that long-term performance, however, an assumption was made that the performance would degrade stepwise after reaching its design life, until the recharge rate matches the rate in the surrounding environment. This approach is based on the assumption that a degraded cover will eventually return to its natural state and will behave like the surrounding environment. A further assumption was that the period of degradation would be the same as the design life. For example, the modified RCRA Subtitle D cover would perform as designed for 100 years and then degrade stepwise in five equal steps over the next 100 years to the point at which recharge rates are equivalent to the rates of the natural surrounding environment.

The schedule and type of engineered cover to be applied to each site was based on the Hanford Disposition Baseline as defined by Kincaid et al. (2000).

Natural (Non-barrier) Recharge Rates

Most of the waste sites at Hanford have not had a surface barrier, and it is assumed that many sites will not have a surface barrier applied prior to Site closure. The effort to estimate recharge in these areas addressed four site conditions:

- undisturbed soil and shrub-steppe vegetation
- undisturbed soil with no vegetation

Table L.6. Barrier Design Lifetimes and Estimated Recharge Rates (actual rates are expected to be less than shown)

DOE-RL Design	Design Life (yr)	Recharge Rate (mm/yr)	Source
Hanford Barrier	1000	0.1	Based on lysimeter data and simulation results (Fayer et al. 1999)
Modified RCRA Subtitle C	500	0.1	Based on lysimeter data and simulation results (Fayer et al. 1999)
Standard RCRA Subtitle C	30	0.1	No data; recommendation is based on presence of geomembrane, 2-ft thick clay admix layer, and short design life
Modified RCRA Subtitle D	100	0.1	Based on simulation results using parameters from Fayer et al. (1999)

- disturbed soil with no vegetation

- disturbed soil with shrub-steppe vegetation.

The Hanford soil map (Hajek 1966) was examined to identify the soil types prevalent in the waste areas. Table L.7 lists the four soil types that dominate the areas being evaluated in the initial assessment and their recharge rates. It was assumed that these soils, in their undisturbed condition, support a shrub-steppe plant community.

For some Hanford activities, the shrub-steppe plant community was often removed while leaving the existing soil type relatively intact. For other activities, the sites were excavated, which removed the existing soil structure, and backfilled with Hanford-formation sand or gravel. Some activities also covered selected surface areas with a layer of gravel (for example, the tank farms). Table L.8 shows the estimated recharge rates for native soils and backfilled sediments without vegetation. Eventually, the disturbed areas may become revegetated and a shrub-steppe plant community re-established. Under these conditions, it is assumed that the estimated recharge rate will return to that equivalent to the pre-Hanford conditions after a period of 100 years.

Summary of Recharge Estimates for the Initial Assessment

The estimated recharge rates for various surface conditions for each of the 13 geographic areas included in the initial assessment are provided in Table L.9. This table presents a brief description of

Table L.7. Estimated Recharge Rates for Predominant Soil Types and Sediments with a Shrub-Step Plant Community

Soil Type	Recharge Rate Estimate (mm/yr)	Description
Ephrata stony loam (Eb)	1.5	No data; used estimate for El, which is a similar soil
Ephrata sandy loam (El)	1.5	Average of two estimates (1.2; 1.8) from deep (> 10 m) chloride data collected from the two boreholes B17 and B18 (Prych 1998)
Burbank loamy sand (Ba)	3.0	Average of three estimates (0.66, 2.8, 5.5) from deep (> 10 m) chloride data collected from the three boreholes B10, B12, and B20 (Prych 1998)
Rupert sand (Rp) inside the 200 East Area	0.9	Average of four estimates (0.16, 0.58, 1.0, and 1.8) from deep (> 10 m) chloride data collected from the four boreholes E24-161, E24-162, B8501, B8502 (Fayer et al. 1999)
Rupert sand (Rp) outside the 200 East Area	4.0	Estimated from chloride data collected from a borehole near the Wye Barricade (Murphy et al. 1996)
Hanford-formation sand	4.0	No data; used estimate for Rupert sand outside the 200 East are

Table L.8. Estimated Recharge Rates for Native Soils and Backfilled Sediments without Vegetation

Soil Type	Recharge Rate Estimate (mm/yr)	Description
Ephrata stony loam (Eb)	17.3	Simulation estimate from Fayer and Walters (1995)
Ephrata sandy loam (El)	17.3	Simulation estimate from Fayer and Walters (1995)
Burbank loamy sand (Ba)	52.5	Simulation estimate from Fayer et al. (1999)
Rupert sand (Rp)	44.3	Simulation estimate from Fayer et al. (1999)
Hanford-formation sand	55.4	8-yr lysimeter record for Hanford sand (Fayer and Walters 1995)
Graveled surface	104	8-yr lysimeter record for graveled surface (Fayer et al. 1999)

each setting and identifies the major soil type that was identified visually for each area using the soil map developed by Hajek (1966). If a substantial secondary soil type was present, that soil type is shown in parentheses. Likewise, its recharge rate is also shown in parentheses. Figure L.2 illustrates how the recharge rates for various surface covers were assumed to change over time, as performance degrades.

The recharge rates estimated for the initial assessment do not account for overland flow from roadways or roofs, water line leaks, or any other anthropogenic additions of water. The rates also do not account for variations within soil types, plant community succession (for example, a takeover by cheatgrass), dune sand deposition, or climate change. Finally, these rates were developed for fairly large geographic areas and may not represent the local recharge rates at specific locations.

L.2.5 Groundwater Module

The Groundwater Module focuses on groundwater that is part of the upper most saturated zone on the Hanford Site. This zone, commonly referred to as the unconfined aquifer, offers a pathway for contaminants released through the vadose zone from past, present, and future site activities to reach the environment accessible to man. Radioactive and hazardous chemicals have been released on the Hanford Site from a variety of sources including ponds, cribs, ditches, injection wells (referred to as reverse wells), surface spills, and tank leaks. Many of these sources have already affected the groundwater, and some may affect it in the future. Once in the groundwater, contaminants move along the pathways of least resistance, from higher to lower potentials, (for example, elevations) where some contaminants may ultimately discharge into the Columbia River.

The goal of the Groundwater Module is to evaluate the transport of contaminants released from the vadose zone to points of regional discharge of groundwater along the Columbia River within the assessment period. Contaminants released to the groundwater form plumes, some of which extend from their source areas to the Columbia River. The Groundwater Module calculates the concentrations of contaminants in the groundwater for direct use in impact and risk calculations.

Table L.9. Recharge estimates for the initial assessment. Substantial secondary soil types and their associated recharge estimates are shown in parentheses.

Area Label	Brief Description	Major (Secondary) Soil Type(s) [(a)]	Recharge Rates Used in the Initial SAC Assessment(s) (mm/yr)			
			Pre- and Post-Hanford (shrub-steppe)	Operations (soil intact, no vegetation)	Operations (soil disturbed, with/without vegetation)	Operations (gravel surface, no vegetation)
C	Reactor along river	Eb (Ba)	1.5 (3.0)	17.3 (52.5)	4.0 / 55.4	104
K	Reactor along river	Eb (El)	1.5 (1.5)	17.3 (17.3)	4.0 / 55.4	104
N	Reactor along river	Eb	1.5	17.3	4.0 / 55.4	104
D	Reactor along river	El	1.5	17.3	4.0 / 55.4	104
H	Reactor along river	Ba	3.0	52.5	4.0 / 55.4	104
F	Reactor along river	Rp (El)	4.0 (1.5)	44.3 (17.3)	4.0 / 55.4	104
R	300 Area	Rp (El)	4.0 (1.5)	44.3 (17.3)	4.0 / 55.4	104
G	200 N Area	El (Ba)	1.5 (3.0)	17.3 (52.5)	4.0 / 55.4	104
T	Northern 200 West Area	Rp (Ba)	4.0 (3.0)	44.3 (52.5)	4.0 / 55.4	104
S	Southern 200 West Area and ERDF	Rp	4.0	44.3	4.0 / 55.4	104
A	Southern 200 East Area	Rp (Ba)	0.9 (3.0)	44.3 (52.5)	4.0 / 55.4	104
B	Northwestern 200 East Area	El	1.5	17.3	4.0 / 55.4	104
E	Eastern 200 East Area	Ba (Rp)	3.0 (0.9)	52.5 (44.3)	4.0 / 55.4	104

Eb = Ephrata stony loam | El = Ephrata sandy loam | Ba = Burbank loamy sand | Rp = Rupert sand

(a) Note: Only the major soil types were used to represent each aggregate area.

Information concerning characterization, modeling, and monitoring of the groundwater system, described in DOE-RL (1999), provides the primary basis for the conceptual model and numerical implementation of the Groundwater Module supporting the initial assessment. The groundwater conceptual model is an interpretation or working description of the characteristics and dynamics of the physical hydrogeologic system, and it consolidates Hanford Site data (for example, geologic, hydraulic, transport, and contaminant data) into a set of assumptions and concepts that can be quantitatively evaluated.

The Groundwater Module takes the results of the analyses from the vadose zone technical element in the form of contaminant flux from various waste sources. In addition to the influx from the vadose zone element, the Groundwater Module requires information that defines the physical characteristics of the hydrologic system, transport parameters, and natural and artificial recharge rates. Driving forces, including natural recharge from precipitation and artificial recharge from waste disposal activities, contribute to

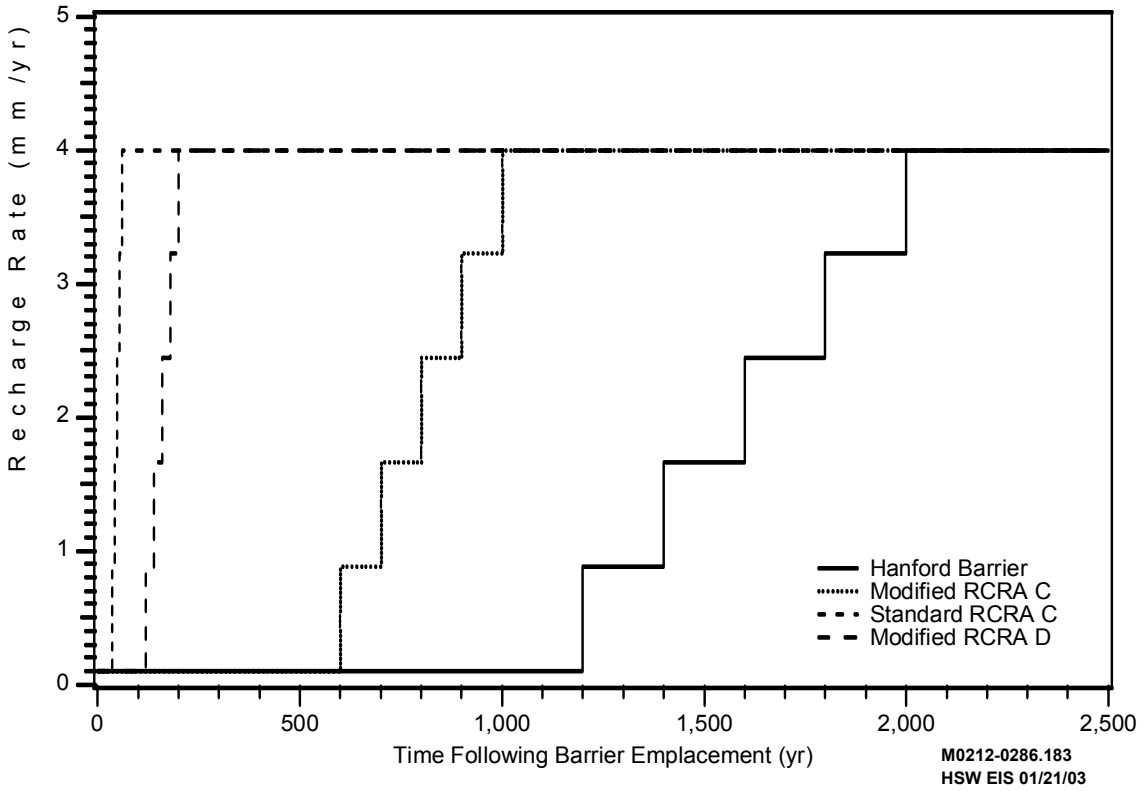

Figure L.2. Recharge Through Covers as a Function of Time

the movement of the contaminants through the vadose zone and into the groundwater of the unconfined aquifer. Several important fate and transport processes, including advection and dispersion, first-order radioactive decay, thermal and chemical interactions with the water and sediment, and contaminant density, may control the fate and transport of the contaminants in the groundwater. For the initial assessment, the thermal and chemical processes considered in the groundwater transport element were limited to assumptions of isothermal conditions, uniform density, and adsorption using the linear sorption isotherm model and, hence, the distribution coefficient, K_d, concept.

The definition of the hydrologic system is based on previous subsurface investigations from which data on the hydrologic units, unit boundaries, hydraulic conductivity, hydraulic heads, storativity, and specific yield were assembled. Transport parameters are based on both site-specific work of previous investigations and published literature values for parameters including effective porosity, dispersivity, contaminant-specific retardation coefficients, and vertical and horizontal anisotropy. The groundwater flow and transport model also requires estimates of natural recharge rates and locations and magnitude of artificial recharge to the hydrologic system, which are available from historic records and direct measurements. Model domain boundaries are established for the flow system based on site-specific knowledge and output data requirements. Boundaries are established along the northern and eastern portion of the Site corresponding to the course of the Columbia River and along the southeastern portion of the model along the course of the Yakima River. Basalt ridgelines and the Cold Creek Valley form the western model domain boundaries. Lower flow boundaries are established between the confined basalt

aquifer system and the overlying unconfined aquifer. A complete description of the groundwater conceptual model is provided in Appendix D of DOE-RL (1999).

The conceptual model of the groundwater system used in this assessment is based on nine major hydrogeologic units identified in Thorne and Chamness (1992), Thorne and Newcomer (1992), and Thorne et al. (1993, 1994). Although nine hydrogeologic units were defined, only seven are found in the unconfined aquifer during the period of interest. The Hanford formation combined with the pre-Missoula gravel deposits were designated as model unit 1. Model units 2 and 3 correspond to the early Palouse soil and Plio-Pleistocene deposits, respectively. Odd-numbered Ringold model units (5, 7, and 9) are predominantly coarse-grained sediment. Even-numbered Ringold model units (4, 6, and 8) are predominantly fine-grained sediment with low permeability. The underlying basalt was designated model unit 10. However, the basalt was assigned a very low hydraulic conductivity and was essentially treated as an impermeable unit in the model.

A complete description of the site-wide groundwater flow and transport model used in the current assessment is provided in Cole et al. (2001a). The current Hanford site-wide groundwater model is implemented with the CFEST code (Gupta et al. 1987). The current model has been transient-inverse calibrated to the record of hydraulic head (that is, water-table elevation) measurements from Hanford startup in 1944 to the present.

Simulated flow conditions during the historical period of operations that provided the basis for all transport calculations are described in Cole et al. (2001b). These flow conditions incorporate the effect of large-volume discharges of wastewater to a variety of waste facilities since the inception of the Hanford Site in 1943. These operational discharges have raised the water table, created groundwater mounds, and been the source of local- and regional-scale contaminant plumes under waste management sites and facilities along the Columbia River and in the Central Plateau. Since 1988, the mission of the Hanford Site has changed from weapons material production to environmental restoration. As a result, wastewater discharges have declined substantially, which caused the water table to decline substantially over the past decade. Simulation of future water table decline indicates that the aquifer would return to more natural levels within 150 to 300 years. These results are consistent with previous work on future water table declines described in Cole et al. (1997) and Kincaid et al. (1998).

The SAC has been inverse calibrated to the hydraulic head data, and history matched to the most abundant data, that for tritium the most mobile of radioactive contaminants. Use of the hydraulic head and tritium data sets provide confidence that the underlying liquid release, vadose zone and groundwater models duplicate the essential features of the tritium groundwater plume; extent of tritium contamination, its arrival at the Columbia River, and its decay as a function of time.

Historical field data specific to solid waste disposal facilities are not available. Solid wastes disposed in containers of either cardboard, wood, plastic, or metal construction are not believed to have released from the their containers and contaminated the sediments immediately below the disposal facilities. It may be decades or centuries before contaminants in some solid waste disposal facilities reach the underlying groundwater and are available for detection. Thus, history matching to solid waste releases is not tractable at this time.

L.2.6 River Transport Module

The River Transport Module simulates the Columbia River between the Vernita Bridge and McNary Dam including inputs from groundwater and the Yakima and the Snake Rivers. The contaminants modeled in the river come from three sources:

- those already in the river when water reaches the Vernita Bridge from upstream sources and atmospheric fallout
- contaminant influx from Hanford waste sites through groundwater
- direct discharge to the river from Hanford facilities.

Groundwater and irrigation return discharges to the river along the shore opposite Hanford are not included in the initial assessment.

The MASS2 code provides the basis of the River Transport Module (Richmond et al. 2000). MASS2 is a two-dimensional, depth-averaged hydrodynamics model that provides the capability to simulate the lateral (bank-to-bank) variation of flow and transport of sediments and contaminants. The model incorporates river hydraulics (velocity and water depth), contaminant influx to the river through groundwater and point sources, sediment and contaminant transport, and adsorption/desorption of contaminant to sediments.

The Columbia River is the largest North American River to discharge into the Pacific Ocean. The river originates in Canada and flows south 1953 km (1212 mi) to the Pacific Ocean. The watershed drains a total of 670,000 km^2 (258,620 mi^2) and receives waters from seven states and one Canadian province. Key contributors to the flow are runoff from the Cascade Mountains in Washington and Oregon and from the western slopes of the Rocky Mountains in Idaho, Montana, and British Columbia. Average annual flows below Priest Rapids and The Dalles dams are approximately 3360 m^3/s (120,000 ft^3/s) and 5376 m^3/s (192,000 ft^3/s), respectively. Numerous dams within the United States and Canada regulate flow on the main stem of the Columbia River. Priest Rapids Dam is the nearest dam upstream of the Hanford Site, and McNary Dam is the nearest downstream. The dams on the lower Columbia River greatly increase the water travel times from the upper reaches of the river to the mouth, subsequently reducing the sediment loads discharged downstream. The increased travel times also allow for greater radionuclide deposition and decay.

The Snake, Yakima, and Walla Walla Rivers all contribute suspended sediment to the Columbia River; contributions from the Snake River are the most substantial. Since completion of McNary Dam in 1953, much of the sediment load has been trapped behind the dam. However, at McNary Dam and other Columbia River dams, some of the trapped sediment is resuspended and transported downstream by seasonal high discharges. As expected, much of this material is redeposited behind dams located farther downstream. Within the domain of this model that only extends to McNary Dam, sediment accumulates faster on the Oregon shore than on the Washington shore because sediment input from the Snake and Walla Walla Rivers stays near the shore on the Oregon side. Sediment-monitoring samples taken for the

Hanford Sitewide Surface Environmental Surveillance Project indicated cobble and coarse- and fine-sand bed sediments at sampling locations along the Hanford Site (Blanton et al. 1995). Silt and clay sediment was observed at the McNary Dam sampling site.

The conceptual model used in the initial assessment included the environmental pathways and transport processes that affect contaminant transport in surface water systems. The physical processes include river hydrodynamics and suspended sediment transport, deposition, and resuspension. Because of run-time constraints, suspended and bed sediments were modeled with only the silt-size fraction. The contaminant transport processes include surface water advection and dispersion, sorption and desorption to sediments, decay, and exchange between bed pore water and the overlying surface water. The initial assessment River Transport Module, which is the MASS2 model, included these key features, events, and processes in the mathematical implementation of the conceptual model.

L.2.7 Risk and Impact

The SAC has implemented a suite of impact assessment modules that treat ecological, economic, cultural, and human impacts and include internal stochastic capabilities. An initial assessment of the Hanford Site using these modules is provided in Bryce et al. (2002). The HUMAN code (Eslinger et al. 2002) was used in calculations for this EIS. The human impact model includes exposure pathways from ingestion, inhalation, skin contact, and direct radiation exposure. Relative exposures to these sources depend on individual lifestyles or exposure scenarios.

The human exposure scenarios for the EIS were limited to the ingestion of water. In addition, the ingestion dose factors were selected as deterministic rather than stochastic factors. With these assumptions, annual human dose calculations do not depend on stochastic variables internal to the human exposure model. Thus, all variability in the human doses arises from the variability in the inventory, release, and transport models. The dose factor used for ingestion of technetium-99 was 1.5×10^{-9} rem/pCi and the dose factor used for ingestion of uranium-238 was 2.5×10^{-7} rem/pCi. These values were obtained by converting the values in Table 2.2 of EPA (1988) from Sv/Bq to rem/pCi (the values were multiplied by a factor of 3700).

Intrusion events by man, vegetation, or animals and the potential for terrestrial ecological pathways to be impacted by Hanford Site wastes in shallow earth deposits is an intrusion analysis – not a long-term exposure analysis. Intrusion analyses are part of the site-specific or waste-specific analyses included in remedial investigation / feasibility studies required under CERCLA, and performance assessment required by DOE Order 435.1. Intrusion analyses contribute to our understanding of the waste concentration that can be safely disposed (i.e., at levels less than chronic and acute intruder dose limits), and of the performance necessary in a barrier system to prevent intrusion by man, vegetation, or animals. However, because intrusion exposures are not included in long-term exposure scenarios, such analyses are not included in the site-wide assessment tool, the SAC.

The version of SAC applied to the initial assessment (Bryce et al. 2002) and that applied in the Hanford SW EIS does not include a terrestrial ecological pathway analysis. Essentially, the SAC does not analyze intruder exposure / risk scenarios. Design of the SAC tool was predicated on the assumption that

the Hanford Site would be closed following the remediation of all sites, and the further assumption that any contaminants at substantial levels in the subsurface would be covered with a proven infiltration and intrusion barrier. A Modified RCRA Subtitle C barrier has been proposed for waste sites receiving surface barriers on the central plateau. Thus, the long-term exposure scenarios do not include intrusion as a source of contamination.

L.2.8 Uncertainty

The SAC was designed to provide a stochastic simulation capability able to quantify uncertainty through a Monte Carlo analysis. An uncertainty analysis can be completed for the SAC results. The goal of such an uncertainty analysis is to determine the model parameters that contribute the most variability to the performance measures. Results of the stochastic realizations can also be used to reveal the maximum – minimum range of performance measures.

The uncertainty analysis addresses the role of uncertainty as caused by the variation of parameters within the modeling systems. It does not address causes of errors between modeled and observed data. It does not address uncertainty due to the use of different models. In addition, the analysis of uncertainty does not differentiate between uncertainties due to lack of knowledge and uncertainty due to natural variability in the parameters.

The uncertainty analysis can identify controlling sources of variability in the simulation estimates of the performance measure, but not necessarily the source of the overall magnitude of the performance measure. However, the source of the overall magnitude is obtained from direct examination of model results.

The uncertainty analysis technique employed is a step-wise linear regression analysis using the output results and input parameters of an assessment. Because the SAC uses a sequential analysis structure (i.e., analysis progressively treats inventory, release, vadose zone, etc.), a top-down hierarchal analysis is performed to identify first tier quantities (e.g., derived quantities like tritium concentration in groundwater), and associated second tier parameters (e.g., unsaturated hydraulic properties, distribution coefficient) responsible for variability.

The initial assessment (Bryce et al. 2002) demonstrated that a relatively small number of input parameter could determine most of the variability in calculated performance measures. It was observed that when the performance measure is human dose, variability with regard to individual behavior and exposure affects uncertainty in the estimated dose more than variability in inventory, release, or environmental transport of the contaminants.

L.3 Results

Results of the initial assessment for a 10,000-year period conducted using the SAC software are presented below in three sections. Section L.3.1 details the release of contamination to the groundwater from the vadose zone. Section L.3.2 presents the drinking water dose that occurs from a 2-L/d drinking water

exposure to groundwater at various points in the environment. Section L.3.3 presents the drinking water dose from consumption of water in the Columbia River at the City of Richland pump station.

L.3.1 Release to Groundwater Results

Releases to the unconfined aquifer from the vadose zone predicted using the SAC software and data are summarized in this section. Vadose zone releases to the groundwater are aggregated into the following categories for the numerous vadose zone sites simulated:

- Solid waste disposal facilities (only '218' sites)
- Tanks (only '241' sites)
- Liquid discharge ('216' sites plus unplanned release sites and the State Approved Land Disposal Site)
- Environmental Restoration Disposal Facility
- Commercial low-level radioactive waste disposal (referred to as the US Ecology site)
- Other sites in 200 East or 200 West Areas not included in the above categories
- All sites not in 200 East or 200 West Areas (that is, 100, 300, 400, and 600 Areas)

For each result, both annual releases and the cumulative of all annual releases (undecayed) are presented. Note, releases from ILAW, melters, and naval reactor compartments are omitted. The stochastic capability of the SAC was employed for these simulations, so the following results are shown in each plot:

- individual stochastic results (25 realizations)
- the median result of the 25 realizations—that is, the realization that resulted in the median cumulative release in the year 12050 A.D. (at the end of the simulation) is emphasized.
- the median-inputs simulation—that is, a separate single-realization simulation with SAC using the median value of all stochastic input variables.

The median result as defined by the cumulative release to the groundwater is highlighted in both the annual release and cumulative release plots. Each new pair of annual and cumulative plots identifies a new median case from the 25 realizations simulated.

The annual release plots have the appearance of being either a series of piecewise constant (stair-step) values or a smooth continuous curve. This is a function of the temporal resolution of both the release model and the vadose zone simulation. Piecewise constant curves result when the release rate is constant

over a period of time and the vadose zone model is able to adopt relatively long time steps (for example, hundreds of years). When either the release or vadose zone model use a fine time step to forecast a more variable release, the release to groundwater appears as a smooth and continuous curve. In reality, both curves are a series of piecewise constant values; however, the fine temporal resolution of the more continuous curve give it the smooth appearance.

Figures L.3 through L.10 present the vadose zone release to groundwater results for the sum of all solid waste disposal facilities. Each cumulative plot showing the 25 stochastic realizations provides information on the range of cumulative response as well as the median for solid waste disposals. Cumulative releases to groundwater for solid waste disposed of in the Central Plateau range from approximately 323 to approximately 445 Ci for technetium-99 during the 10,000-year analysis period. However, for uranium the release is nil—none in any realization in the 200 East Area and only 5 of 25 realizations exhibit any release in 200 West Area. The median solutions for both 200 East and 200 West Areas are zero essentially.

Figures L.11 through L.18 present the results for vadose zone releases to groundwater for the sum of all tank sites. Cumulative releases to groundwater for tank waste (that is, past leaks, future losses, and residuals) in the Central Plateau range from approximately 440 to approximately 645 Ci for technetium-99 during the 10,000-year analysis period. As in the case of solid waste, uranium in tank waste does not exhibit substantial release during the 10,000-year period. Only 5 of 25 realizations show uranium release from 200 East Area tank sites, and hence, the median release is zero. For 200 West Area tank sites, the median case predicts release of approximately 1 Ci of uranium to groundwater during the entire 10,000-year period.

Figures L.19 through L.26 present the vadose zone release to groundwater results for the sum of all liquid discharge and unplanned release (UPR) sites and (in the case of 200 West) the SALDS facility. Cumulative releases to groundwater for liquid releases in the Central Plateau range from approximately 735 to approximately 1030 Ci for technetium-99 during the 10,000-year analysis period. The vast majority of this activity is associated with 200 East Area. The liquid release of uranium ranges between approximately 5 and approximately 100 Ci for the Central Plateau with median values of approximately 26 Ci for 200 East Area and approximately 5 Ci for 200 West Area.

Figures L.27 through L.38 present the results for vadose zone releases to groundwater for the sum of all other sites (sites in 200 East and 200 West Areas, excluding solid waste burial ground, tank, liquid discharge, unplanned release, ERDF, and commercial low-level radioactive waste disposal sites) and for the sum of all sites outside the 200 East and 200 West Areas (that is, the 100, 300, 400, and 600 area sites). Cumulative releases to groundwater for all other sites (for example, canyons, tunnels) on the Central Plateau range from approximately 15 to approximately 50 Ci for technetium-99 during the 10,000-year analysis period. The majority of this activity is associated with 200 West Area. Negligible releases of uranium occur from these sites. Cumulative releases to groundwater from sites away from the Central Plateau (for example, river corridor sites with residual contamination) range from approximately 17 to approximately 37 Ci for technetium-99 during the 10,000-year analysis period. The release of uranium from these same sites ranges from approximately 5 to approximately 80 Ci. Note that the river corridor

includes several liquid waste disposal trenches that received fuel fabrication waste streams that carried uranium to the vadose zone.

Figures L.39 through L.42 present the results for vadose zone releases to groundwater for the ERDF. Cumulative releases to groundwater from the ERDF range from 0 to approximately 27 Ci for technetium-99 during the 10,000-year analysis period. As in the case of solid waste, uranium in the ERDF does not exhibit significant release during the 10,000-year period. Only 3 of 25 realizations exhibit any release, none before 7000 years post-closure. Hence, the median case shows no uranium release to groundwater.

Figures L.43 through L.46 present the results for vadose zone releases to groundwater for the commercial low-level radioactive waste disposal site operated by US Ecology, Inc. Cumulative releases to groundwater from the US Ecology site range from 0 to approximately 80 Ci for technetium-99 during the 10,000-year analysis period. The annual release curves (Figure L.43) and the cumulative plots (Figure L.44) exhibit substantial variability in the timing of release; however, the peak annual releases appear to vary between only approximately $2x10^{-2}$ and approximately $5x10^{-2}$ Ci/yr after 3000 A.D. As in the case of solid waste and ERDF, uranium in the US Ecology site does not exhibit release to groundwater during the 10,000-year period.

These results indicate that technetium-99 releases from the solid waste disposal facilities to groundwater of may account for approximately 323 to approximately 445 Ci in 10,000 years, and releases of uranium would be negligible. This contrasts with approximately 440 to approximately 645 Ci of technetium-99 from tank sites, approximately 735 to approximately 1030 Ci from liquid releases, approximately 15 to approximately 50 Ci from other sites on the Central Plateau, approximately 17 to approximately 37 Ci from sites away from the plateau, 0 to approximately 27 Ci from ERDF, and 0 to approximately 80 Ci from the US Ecology site. Overall, the comparison is approximately 323 to approximately 445 Ci of technetium-99 from solid waste and approximately 1530 to approximately 2310 Ci of technetium-99 released in 10,000 years from all Hanford Site sources. Thus, the contribution from Hanford solid waste would amount to about 20 percent of the cumulative technetium-99 release from all Hanford sources.

The release of uranium to groundwater from Hanford solid waste is much lower. No realizations showed any release of uranium to groundwater from Hanford solid waste in the 200 East Area, and only 5 of 25 realizations exhibit any release of uranium to groundwater from Hanford solid waste in 200 West Area. Thus, in an average, or median, sense, Hanford solid waste deposits would release no uranium to groundwater over the 10,000-year period of analysis. This result compares to a median release of approximately 84 Ci and a range of release to groundwater from the 25 realizations of between approximately 10 and approximately 300 Ci of uranium for all Hanford wastes. Of the five realizations of non-zero uranium release from Hanford solid waste in the 200 West Area, the range of cumulative release was 0 to approximately 94 Ci. Hence, the contribution to overall uranium release to the water table from Hanford solid waste lies between 0 and approximately 29 Ci, but the majority of realizations show zero release. As a consequence, the contribution from Hanford solid waste would amount to between 0 and 30 percent of the cumulative release from all Hanford sources. The majority of the technetium-99 and uranium release was forecast to occur from past liquid discharge sites (cribs, ponds, trenches) and unplanned releases on the plateau, and from off-plateau or river corridor waste sites.

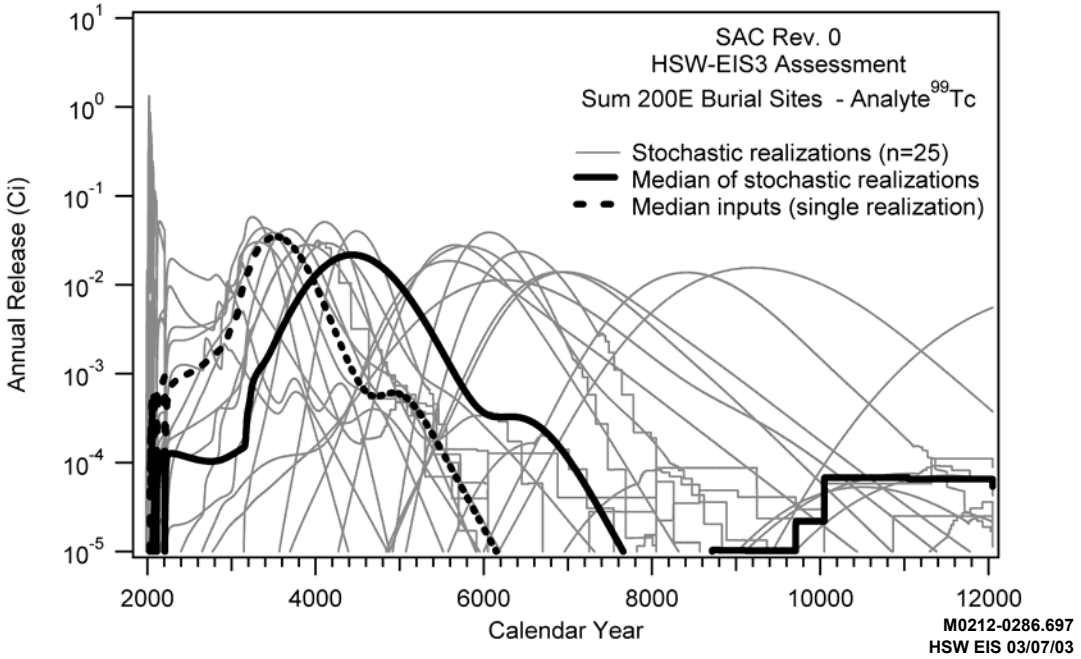

Figure L.3. SAC Results for Annual Vadose Zone Release of Technetium-99 from All Solid Waste Disposal Facilities Sites in the 200 East Area (including all '218' sites except 218-E-14 and 218-E-15, and excluding ILAW)

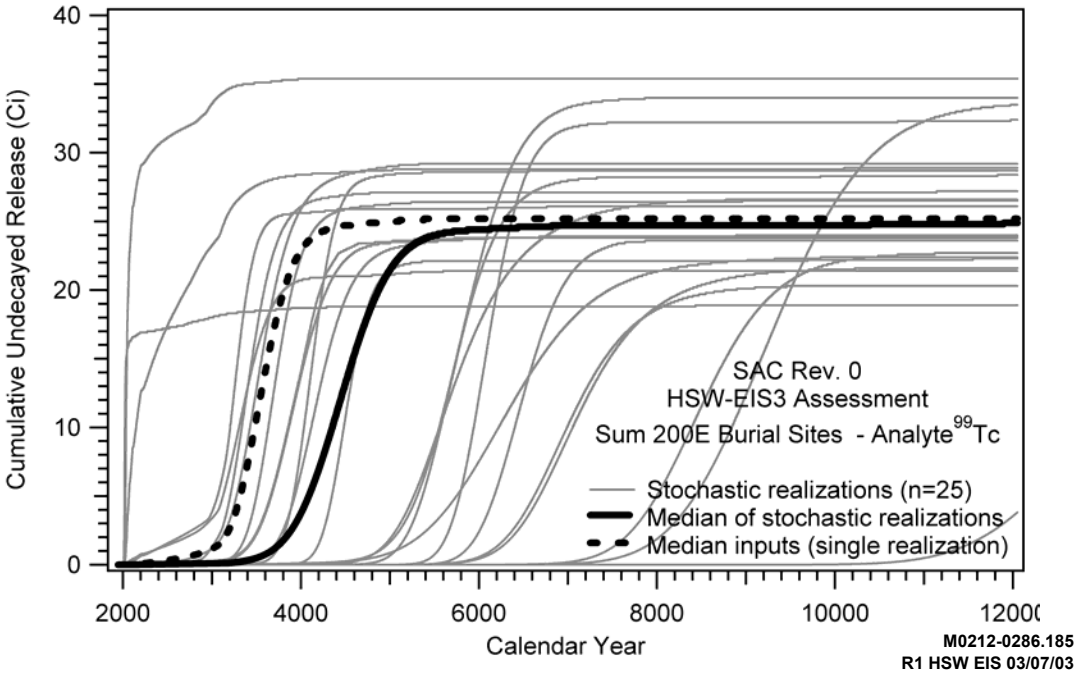

Figure L.4. SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Solid Waste Disposal Facilities Sites in the 200 East Area (including all '218' sites except 218-E-14 and 218-E-15, and excluding ILAW)

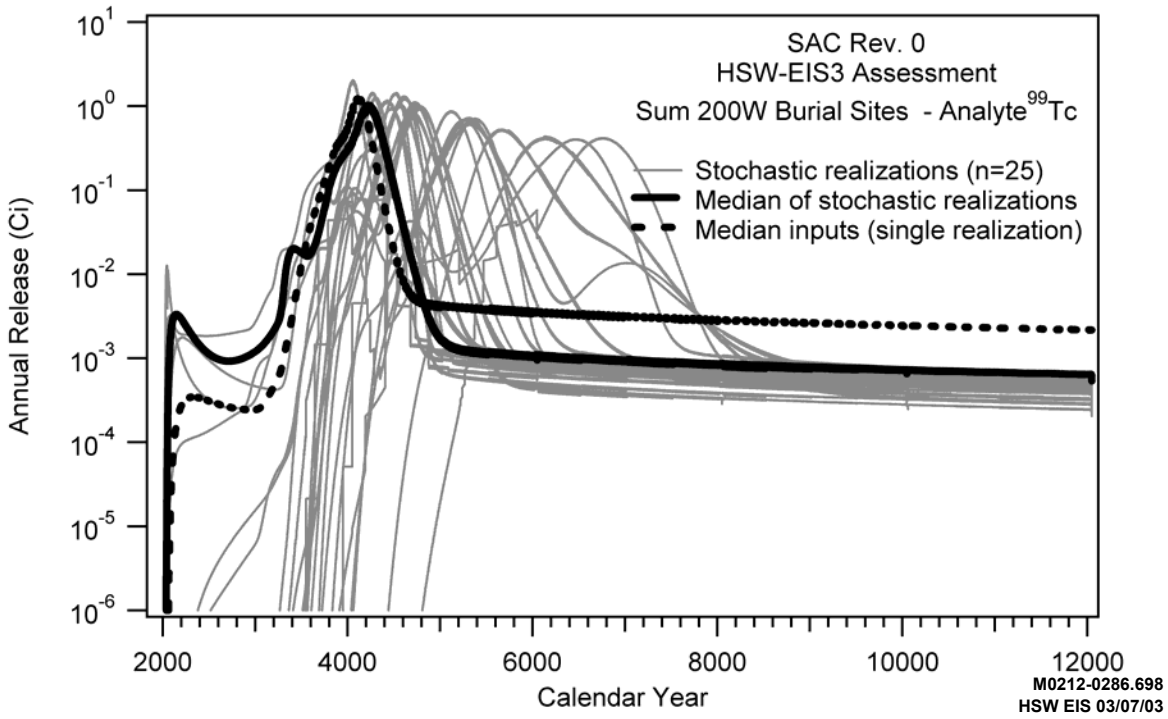

Figure L.5. SAC Results for Annual Vadose Zone Release of Technetium-99 from All Solid Waste Disposal Facilities Sites in the 200 West Area (including all '218' sites)

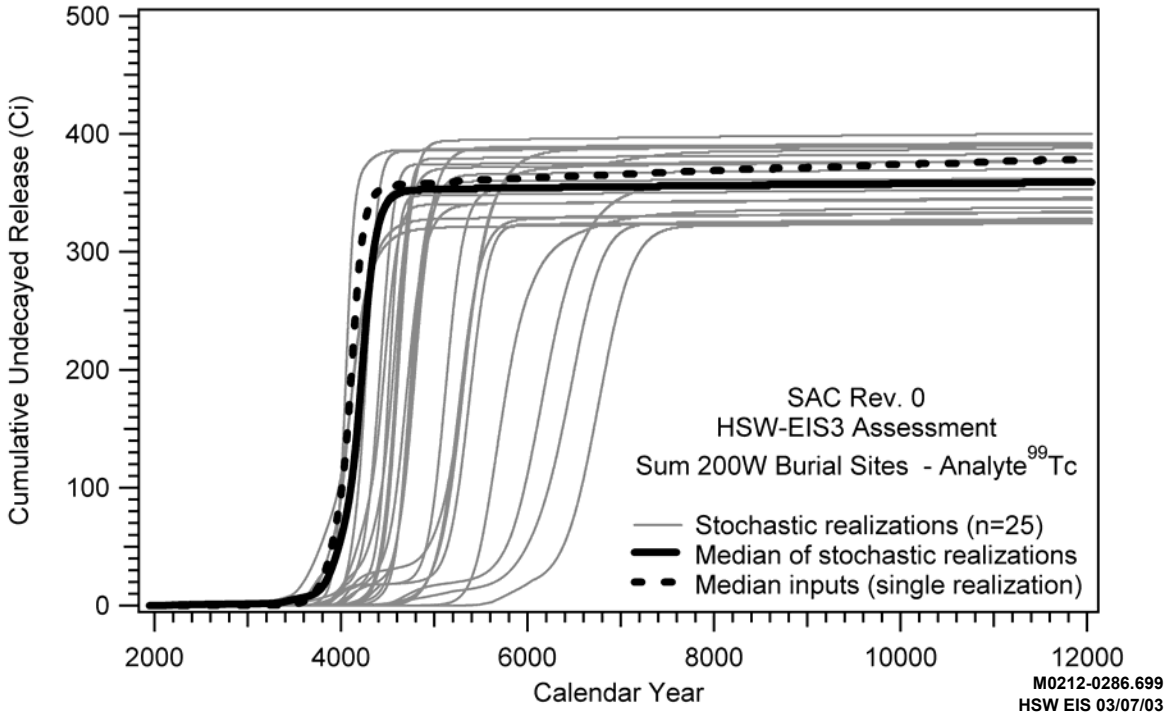

Figure L.6. SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Solid Waste Disposal Facilities Sites in the 200 West Area (including all '218' sites)

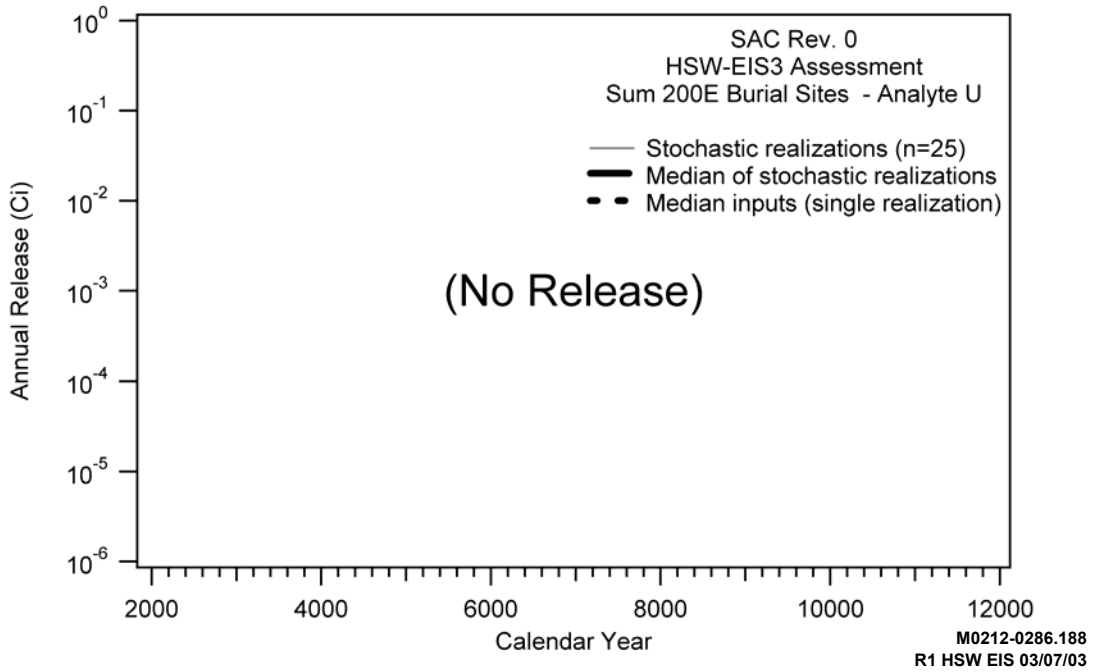

Figure L.7. SAC Results for Annual Vadose Zone Release of Uranium from All Solid Waste Disposal Facilities Sites in the 200 East Area (including all '218' sites except 218-E-14 and 218-E-15, and excluding ILAW)

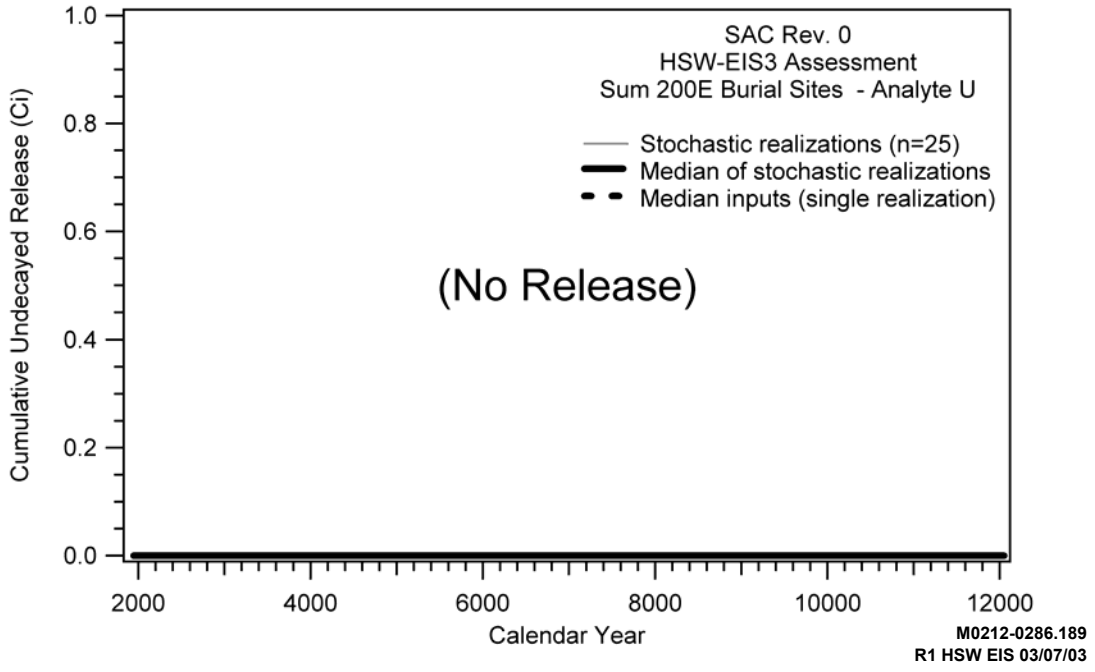

Figure L.8. SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Solid Waste Disposal Facilities Sites in the 200 East Area (including all '218' sites except 218-E-14 and 218-E-15, and excluding ILAW)

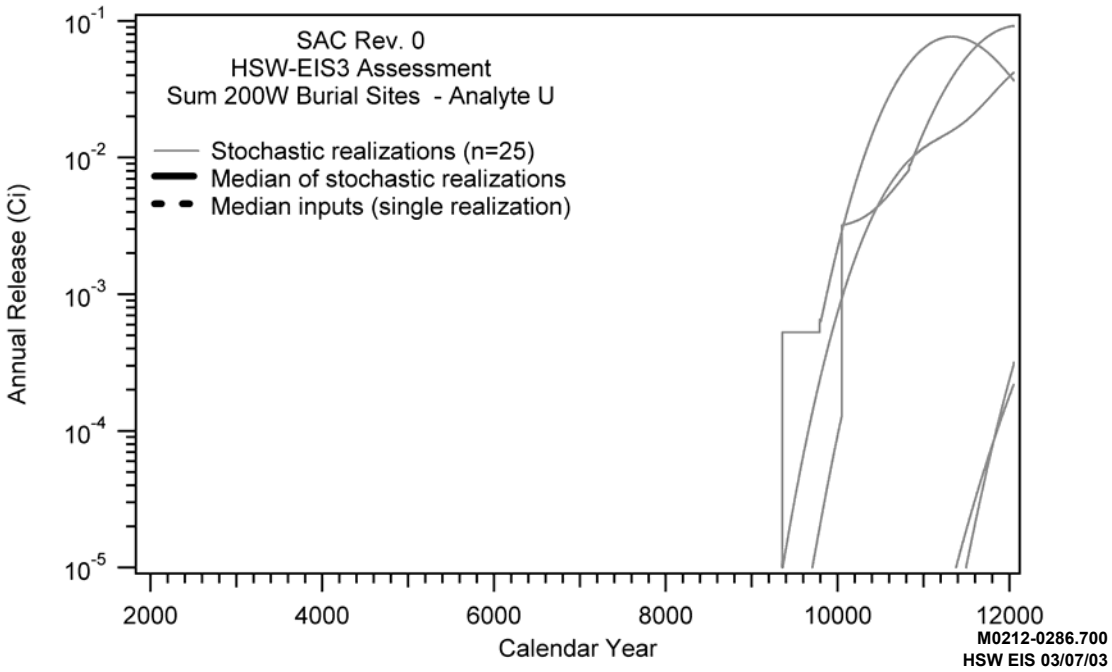

Figure L.9. SAC Results for Annual Vadose Zone Release of Uranium from All Solid Waste Disposal Facilities Sites in the 200 West Area (including all '218' sites)

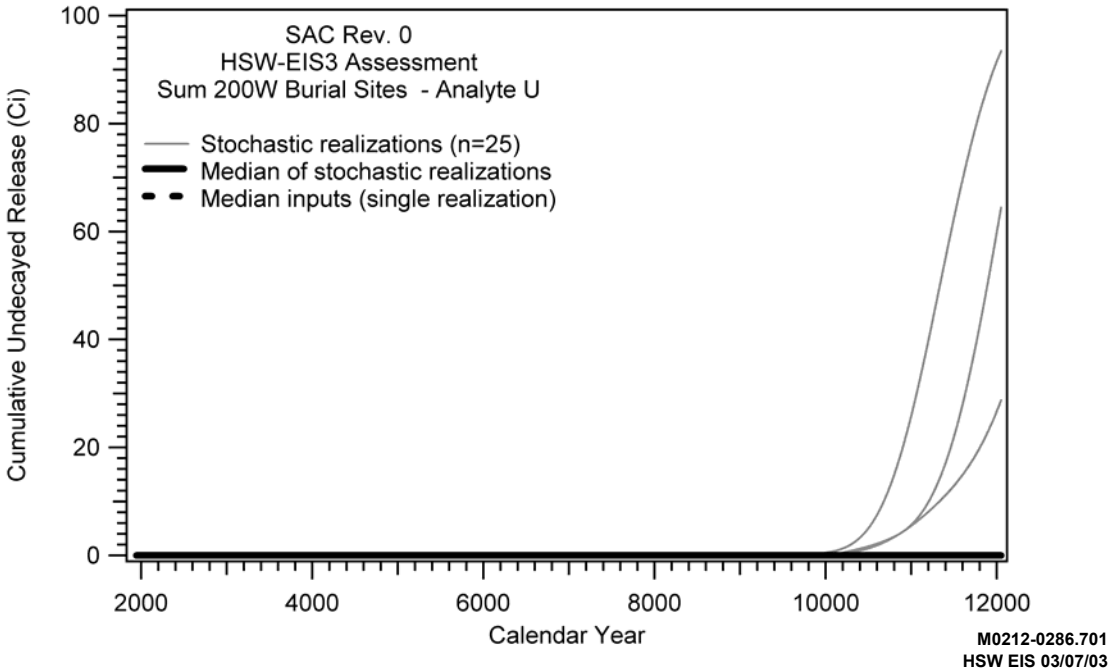

Figure L.10. SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Solid Waste Disposal Facilities Sites in the 200 West Area (including all '218' sites)

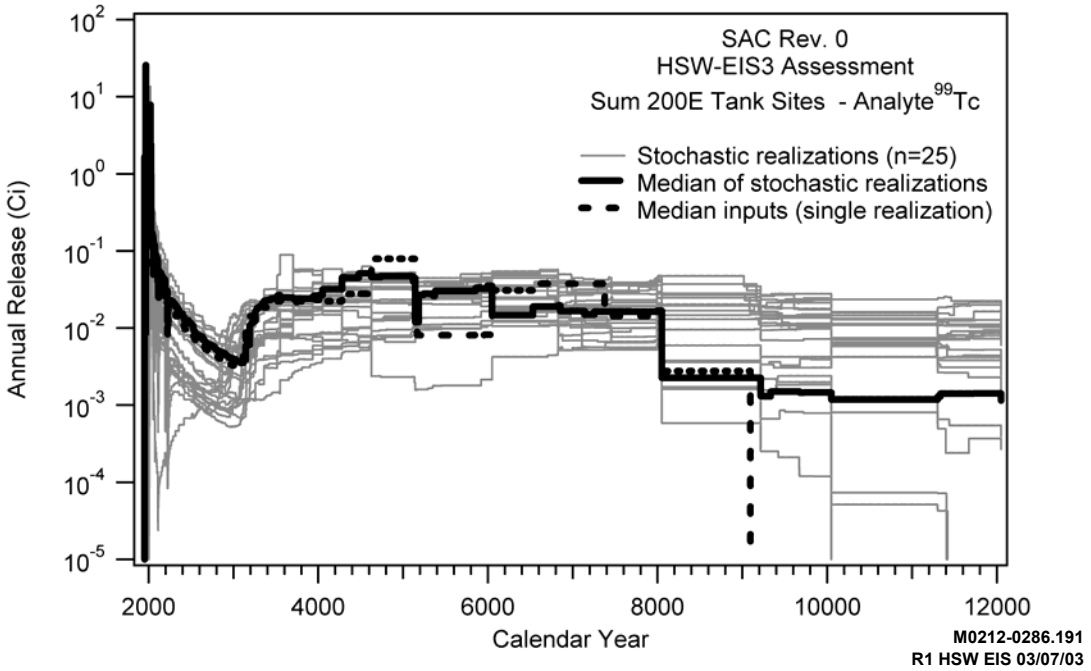

Figure L.11. SAC Results for Annual Vadose Zone Release of Technetium-99 from All Tank Sites in the 200 East Area

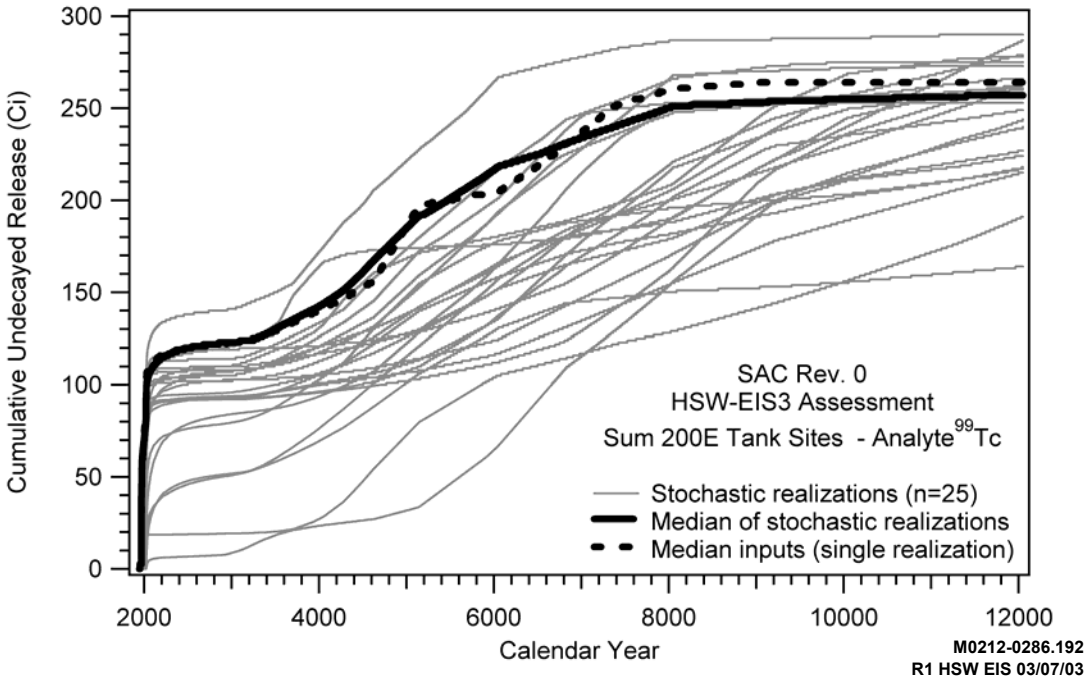

Figure L.12. SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Tank Sites in the 200 East Area

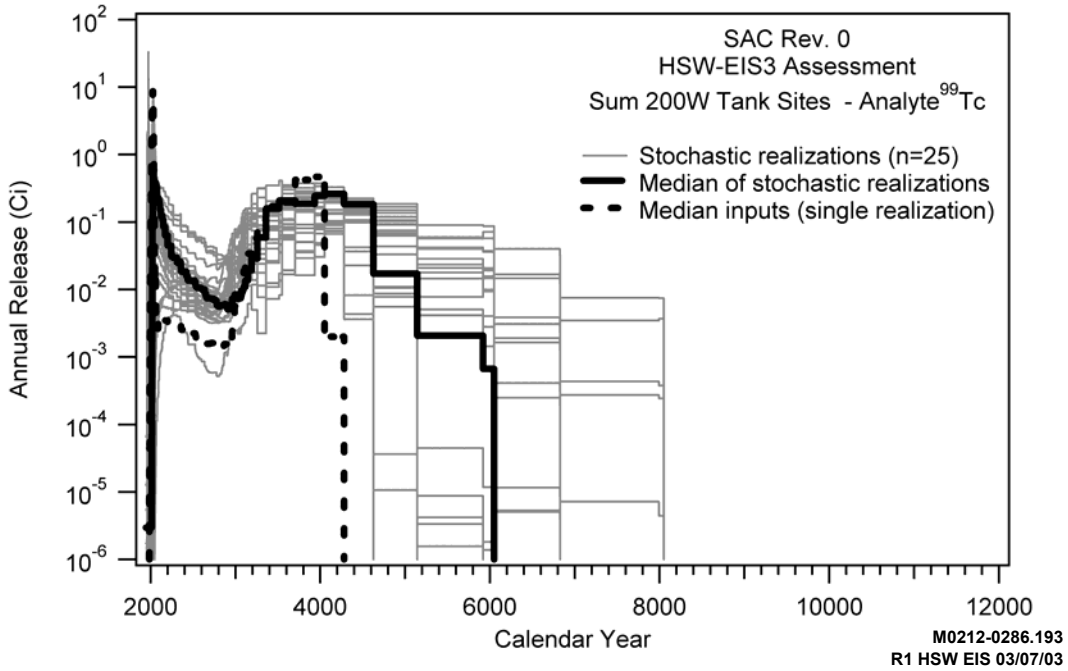

Figure L.13. SAC Results for Annual Vadose Zone Release of Technetium-99 from All Tank Sites in the 200 West Area

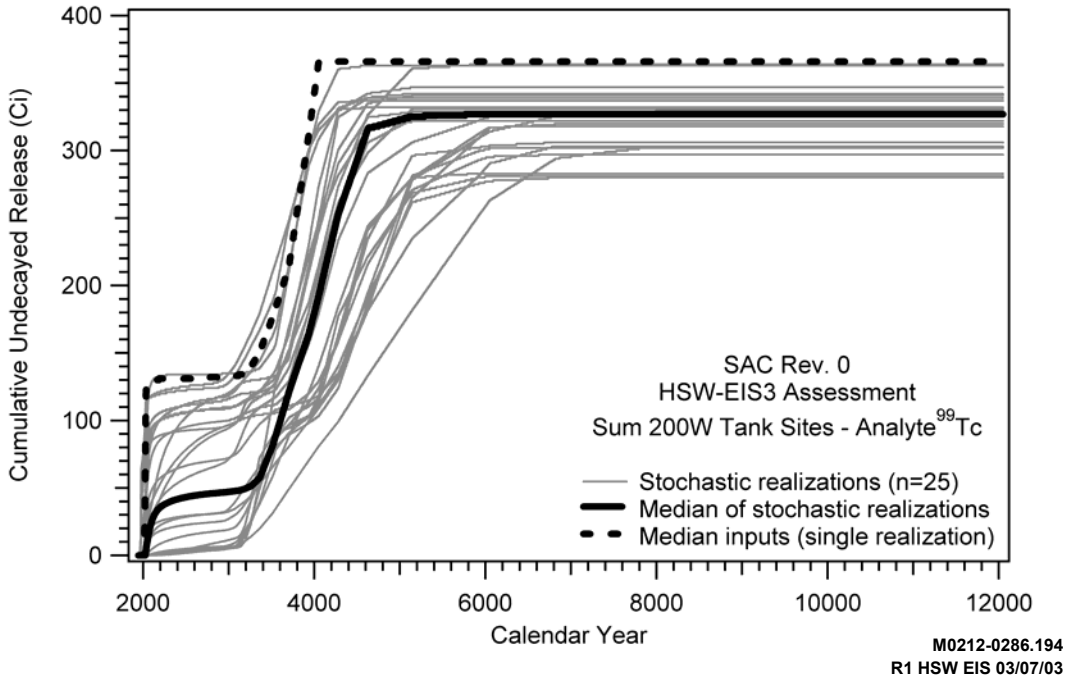

Figure L.14. SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Tank Sites in the 200 West Area

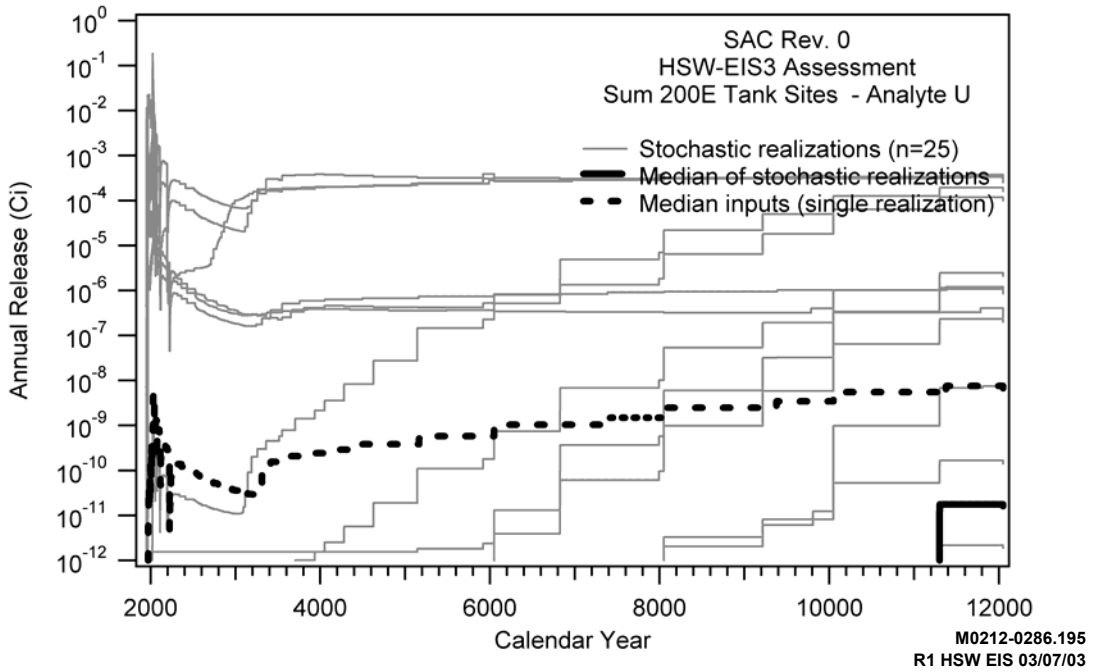

Figure L.15. SAC Results for Annual Vadose Zone Release of Uranium from All Tank Sites in the 200 East Area

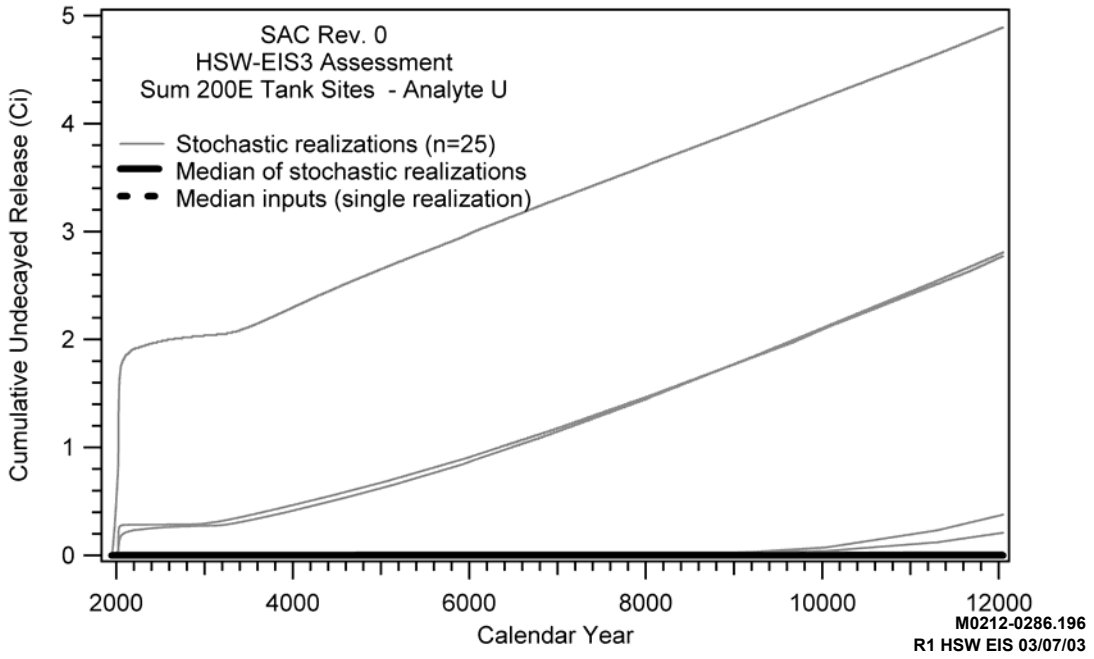

Figure L.16. SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Tank Sites in the 200 East Area

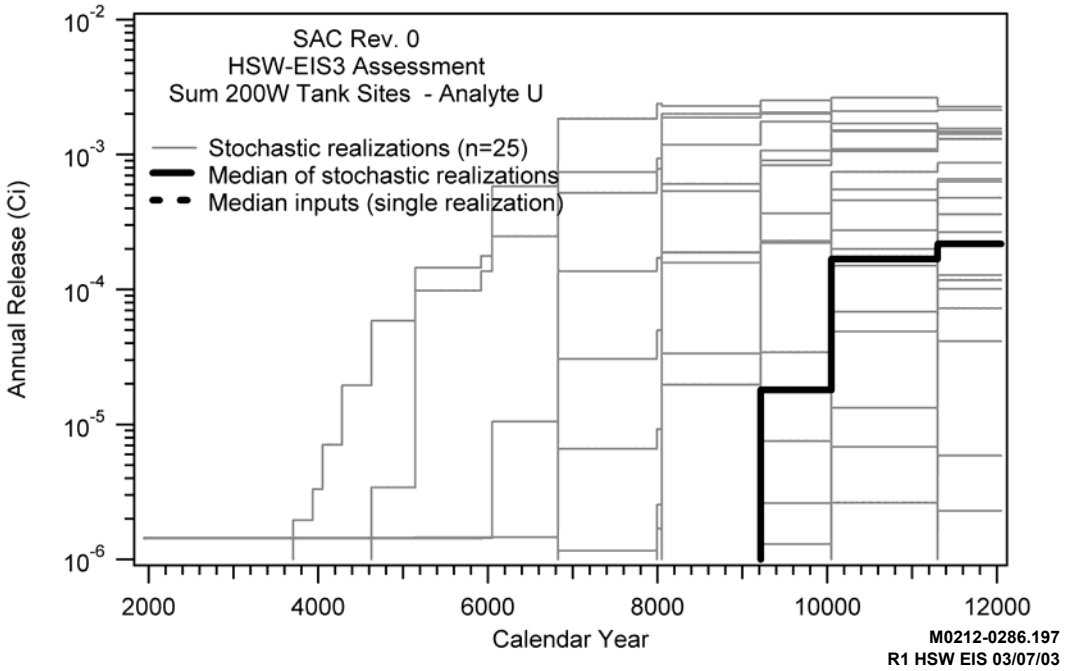

Figure L.17. SAC Results for Annual Vadose Zone Release of Uranium from All Tank Sites in the 200 West Area

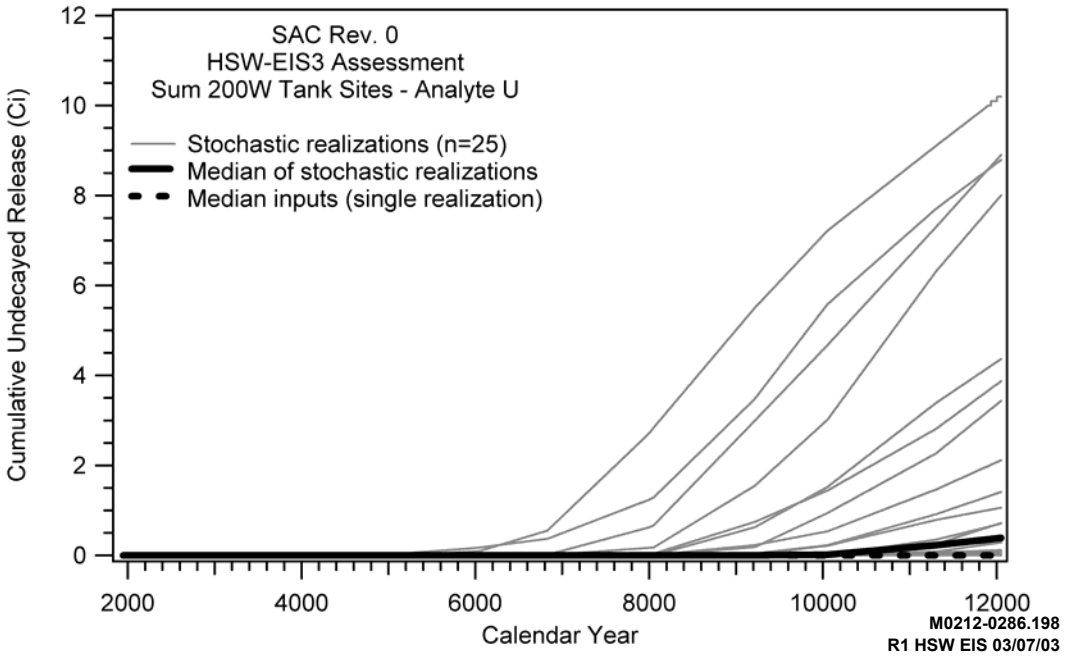

Figure L.18. SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Tank Sites in the 200 West Area

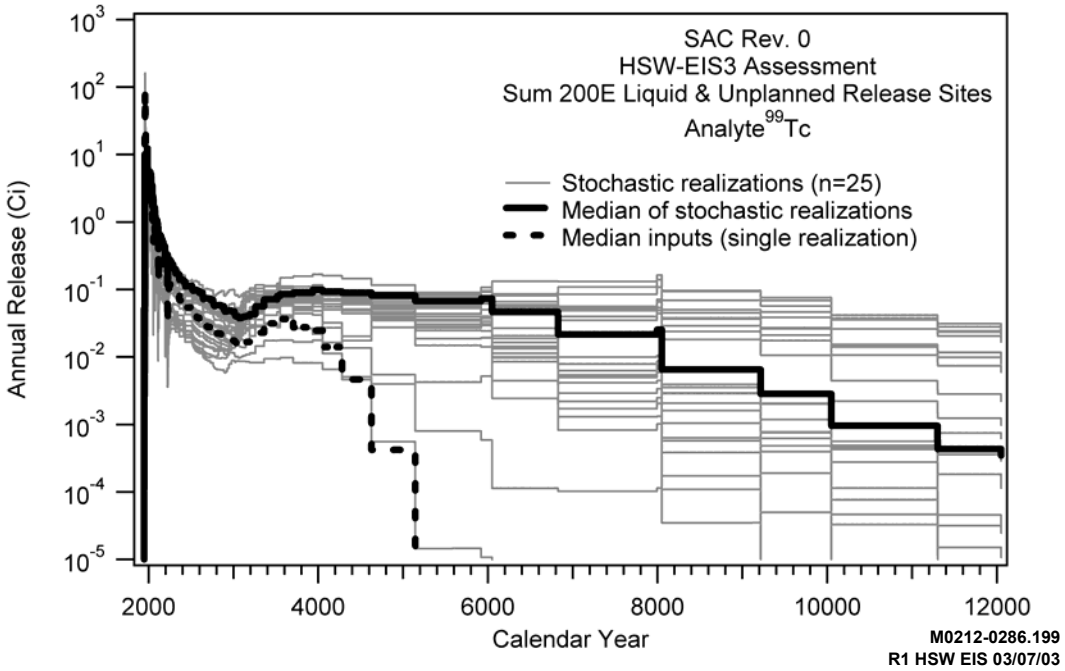

Figure L.19. SAC Results for Annual Vadose Zone Release of Technetium-99 from All Liquid Discharge and Unplanned Release Sites in the 200 East Area

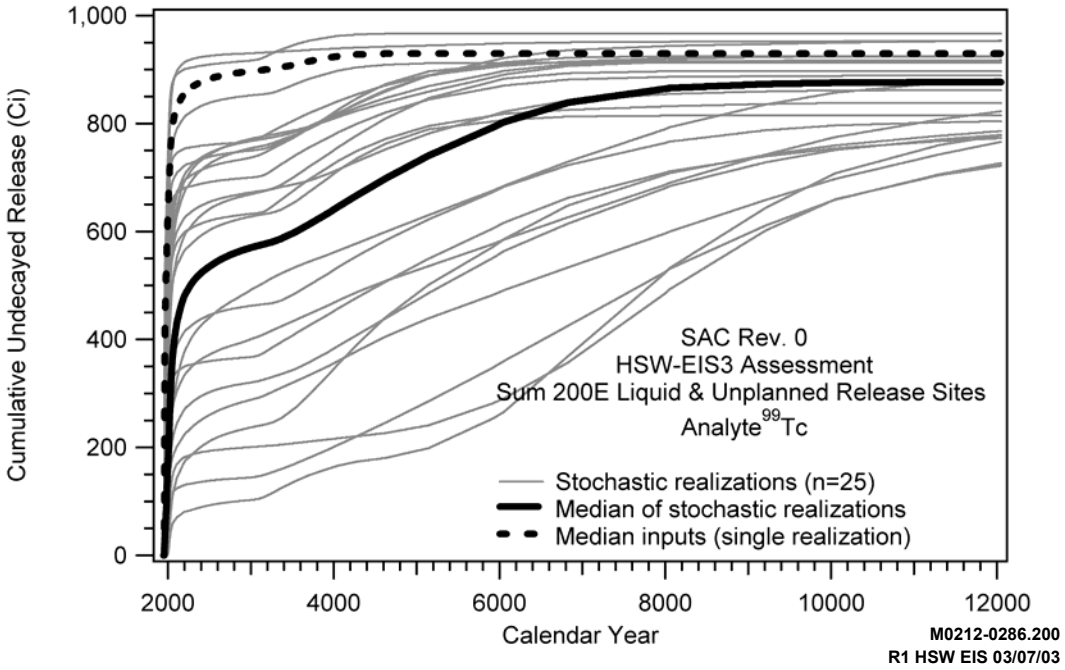

Figure L.20. SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Liquid Discharge and Unplanned Release Sites in the 200 East Area

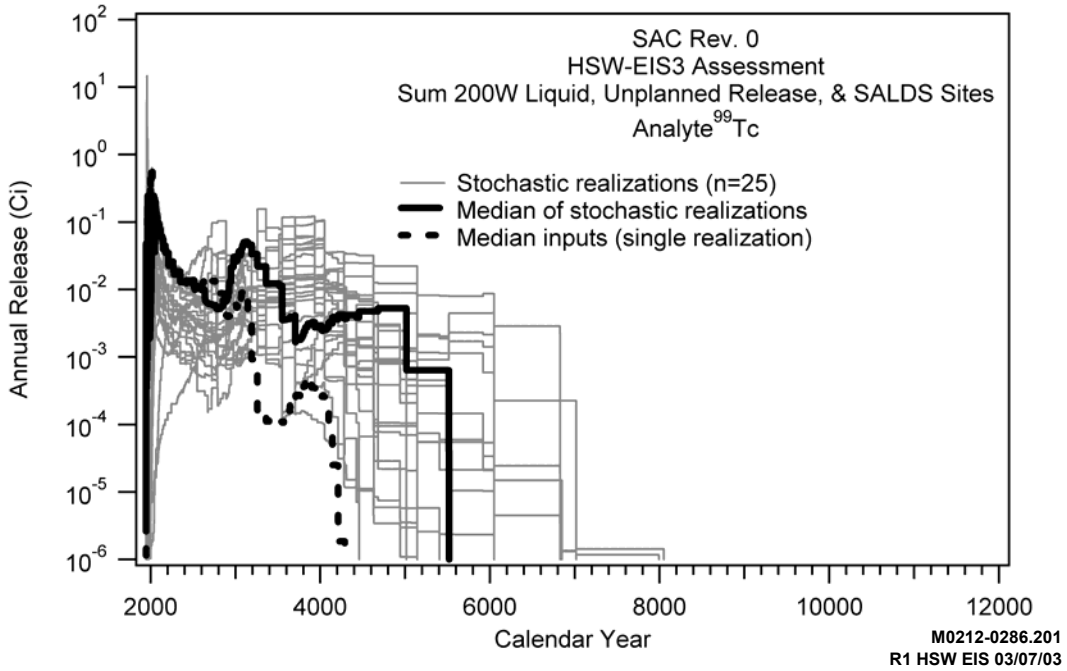

Figure L.21. SAC Results for Annual Vadose Zone Release of Technetium-99 from All Liquid Discharge and Unplanned Release Sites in the 200 West Area Plus SALDS

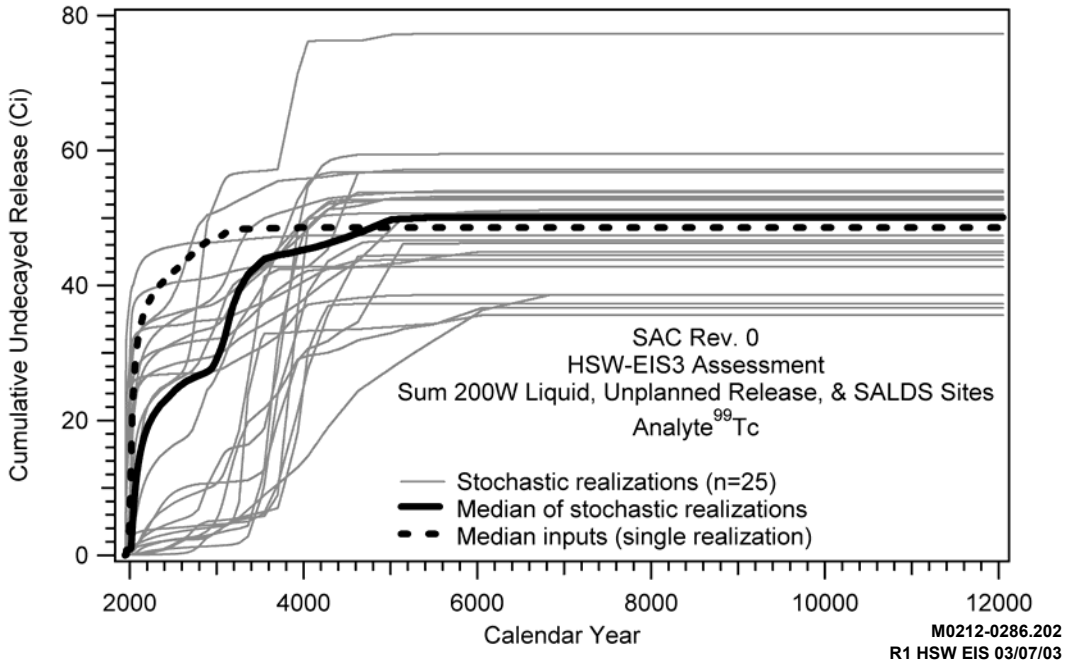

Figure L.22. SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Liquid Discharge and Unplanned Release Sites in the 200 West Area Plus SALDS

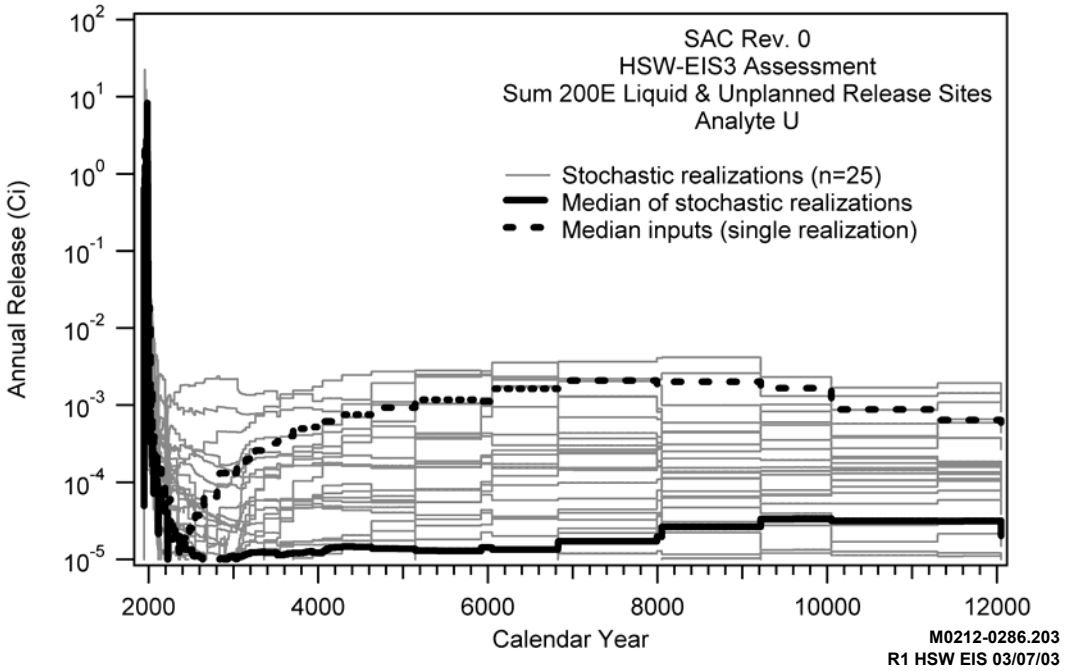

Figure L.23. SAC Results for Annual Vadose Zone Release of Uranium from All Liquid Discharge and Unplanned Release Sites in the 200 East Area

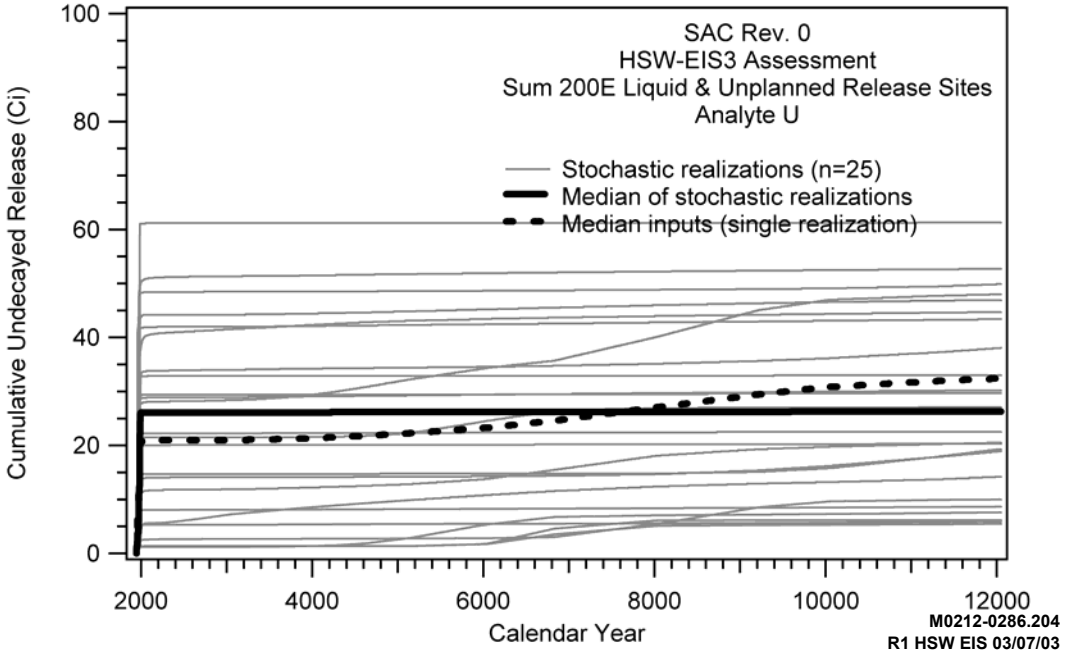

Figure L.24. SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Liquid Discharge and Unplanned Release Sites in the 200 East Area

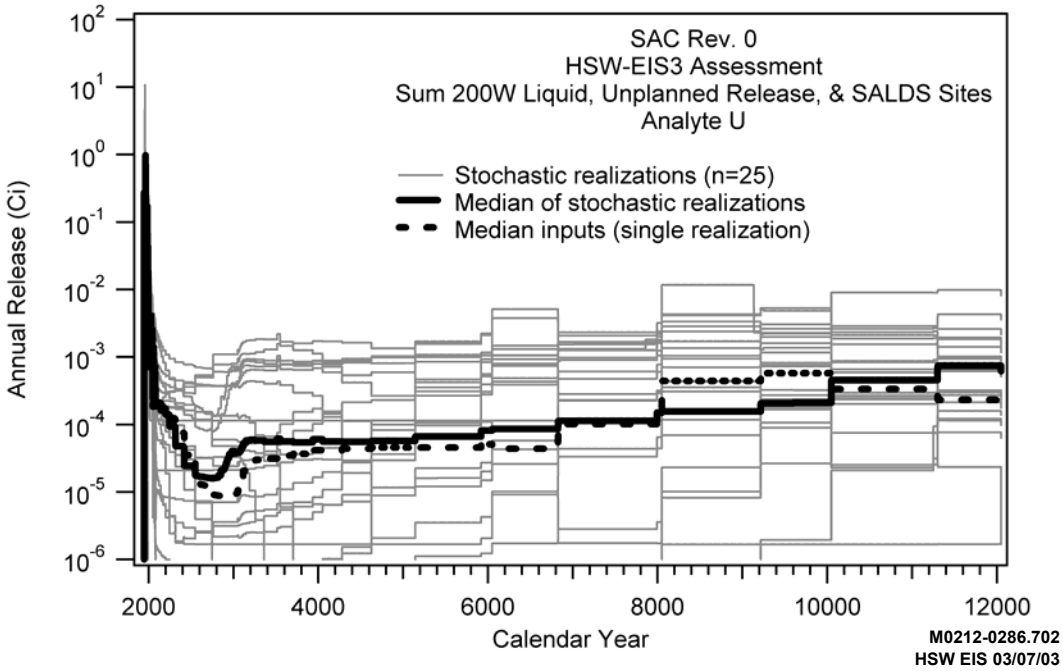

Figure L.25. SAC Results for Annual Vadose Zone Release of Uranium from All Liquid Discharge and Unplanned Release Sites in the 200 West Area Plus SALDS

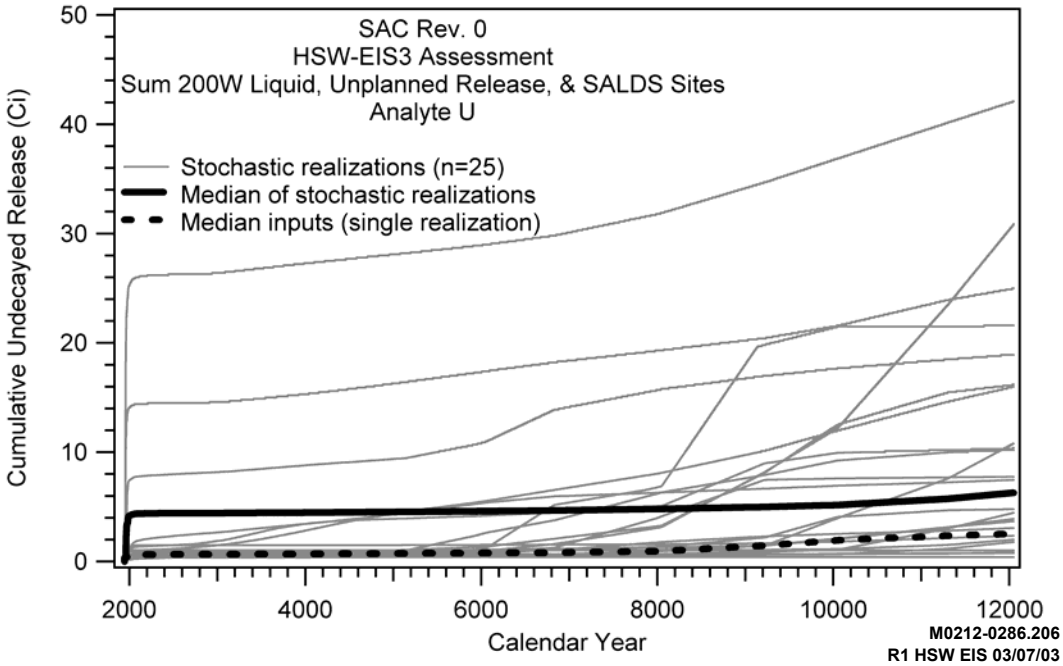

Figure L.26. SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Liquid Discharge and Unplanned Release Sites in the 200 West Area Plus SALDS

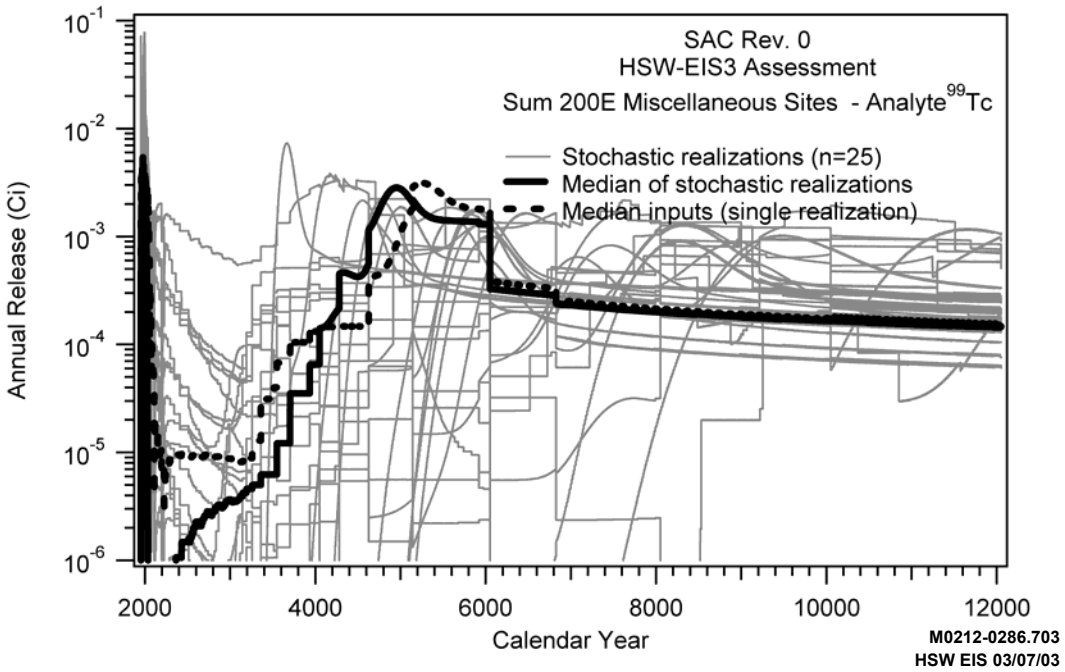

Figure L.27. SAC Results for Annual Vadose Zone Release of Technetium-99 from All Other Sites in the 200 East Area

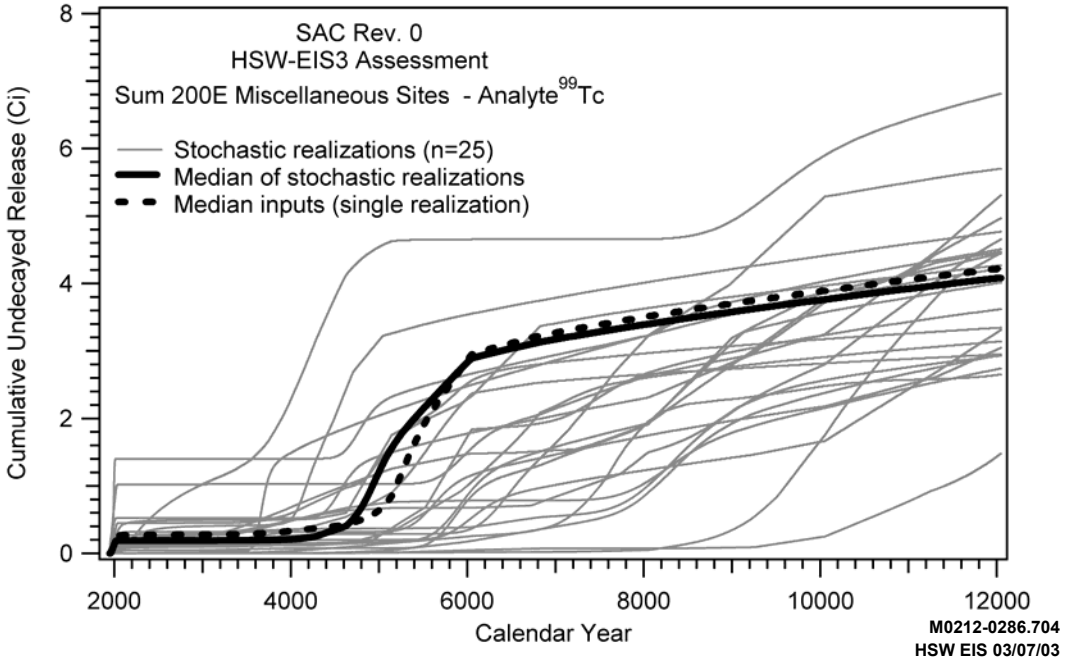

Figure L.28. SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Other Sites in the 200 East Area

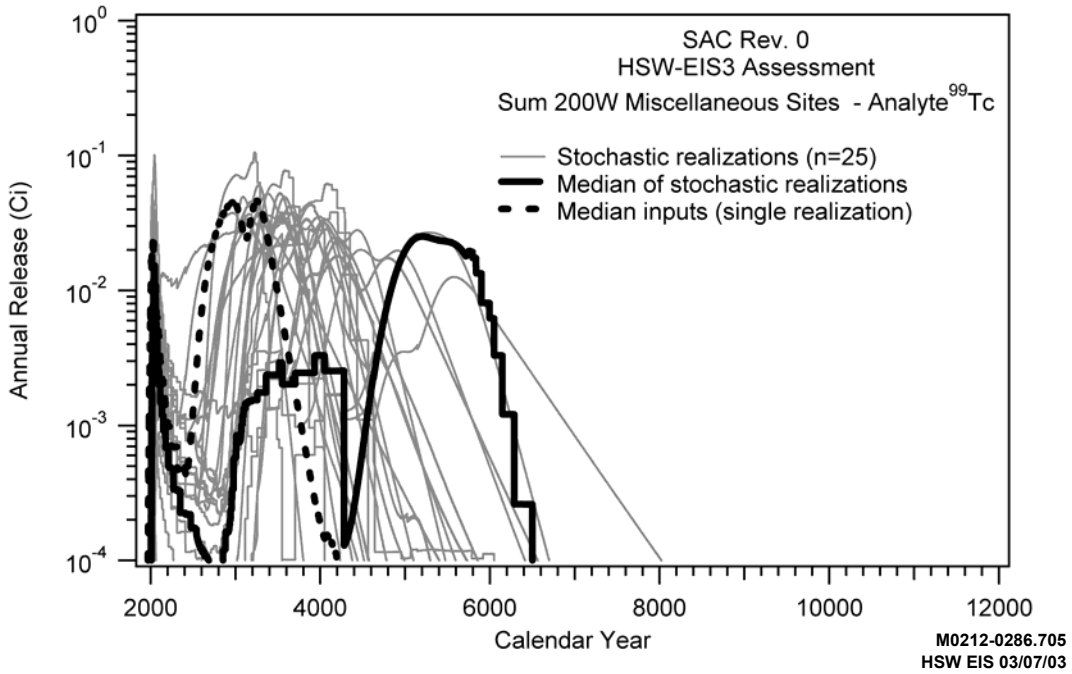

Figure L.29. SAC Results for Annual Vadose Zone Release of Technetium-99 from All Other Sites in the 200 West Area

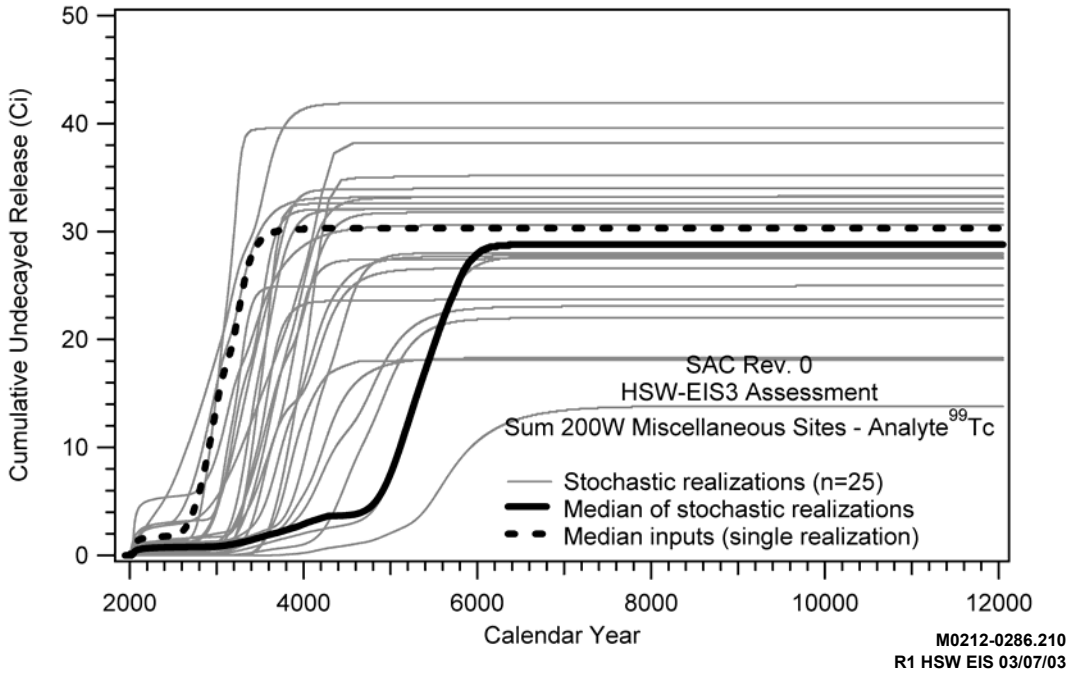

Figure L.30. SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Other Sites in the 200 West Area

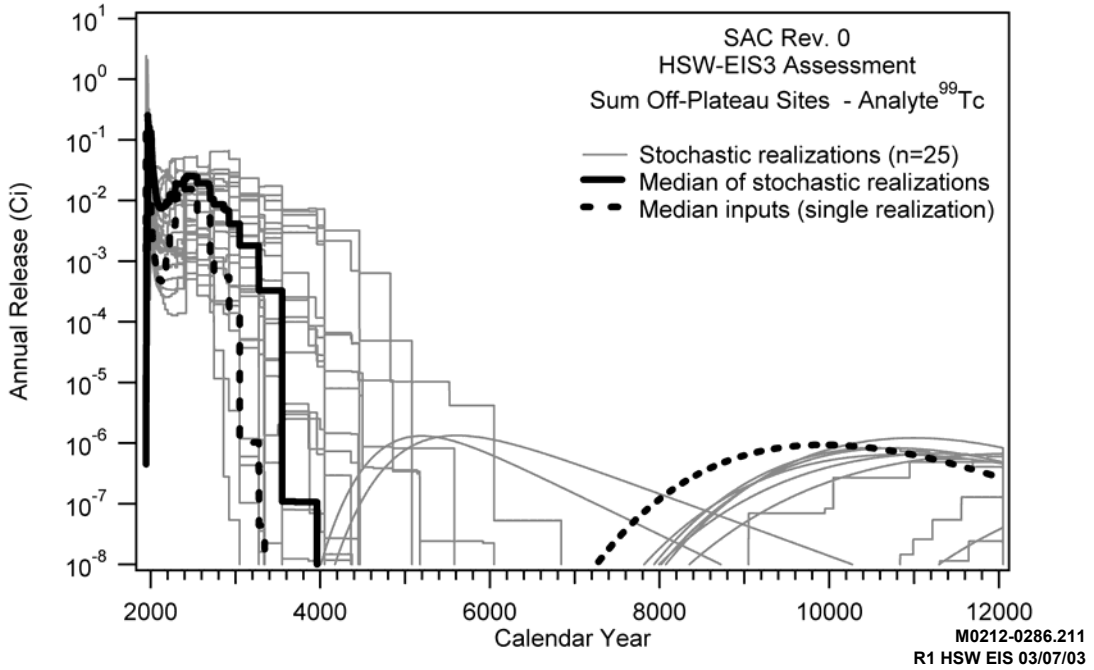

Figure L.31. SAC Results for Annual Vadose Zone Release of Technetium-99 from all Other Sites Outside the 200 East and 200 West Areas

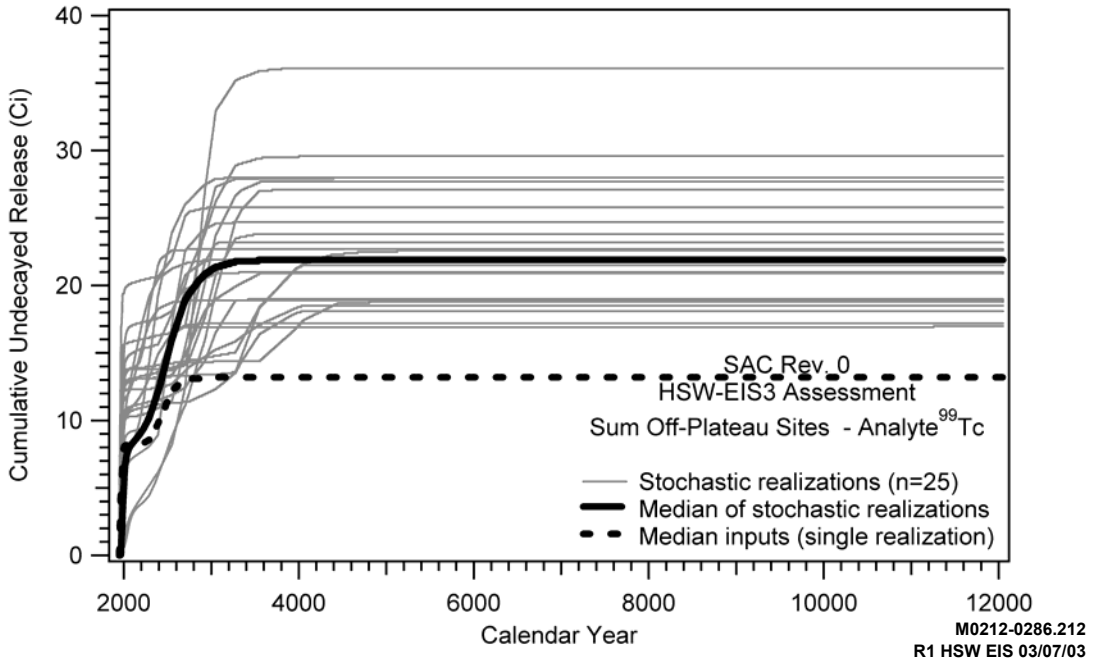

Figure L.32. SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Other Sites Outside the 200 East and 200 West Areas

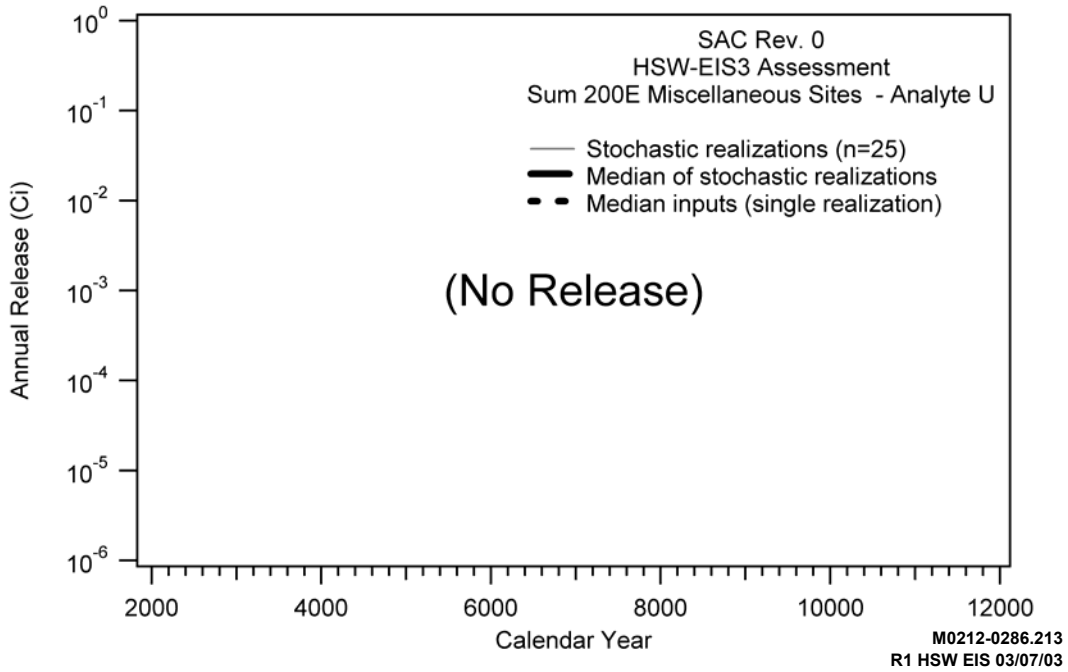

Figure L.33. SAC Results for Annual Vadose Zone Release of Uranium from All Other Sites in the 200 East Area

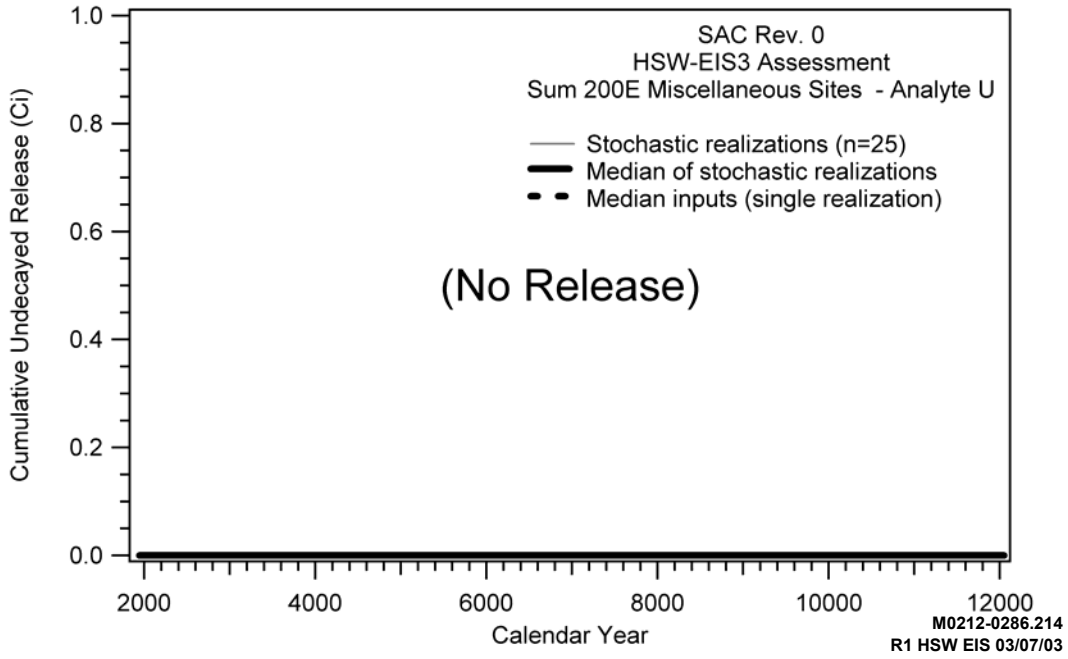

Figure L.34. SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Other Sites in the 200 East Area

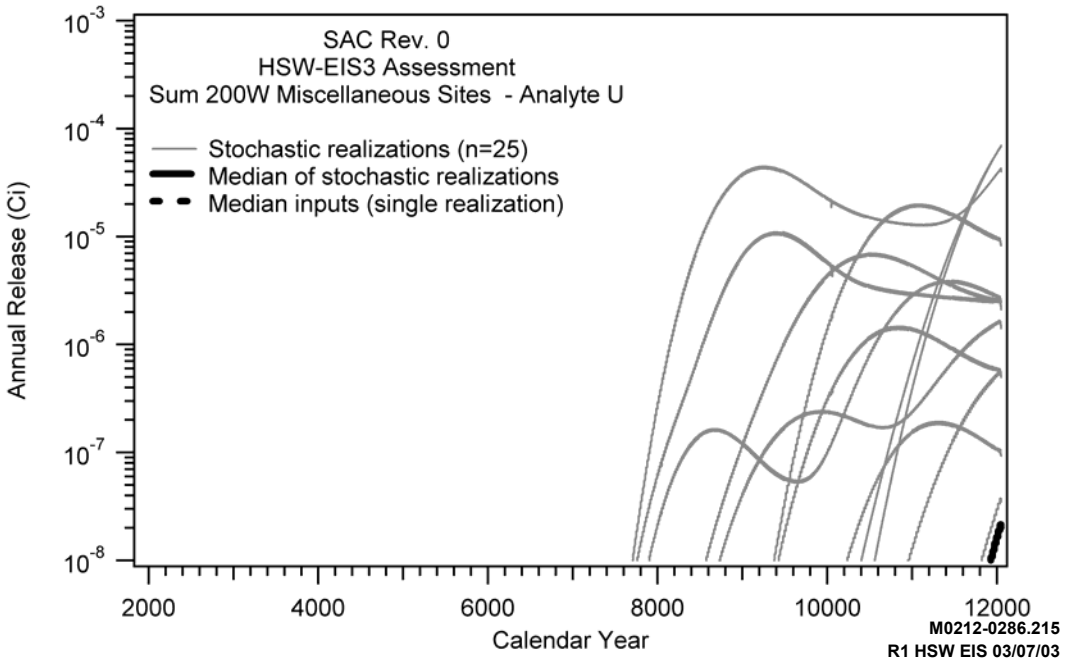

Figure L.35. SAC Results for Annual Vadose Zone Release of Uranium from All Other Sites in the 200 West Area

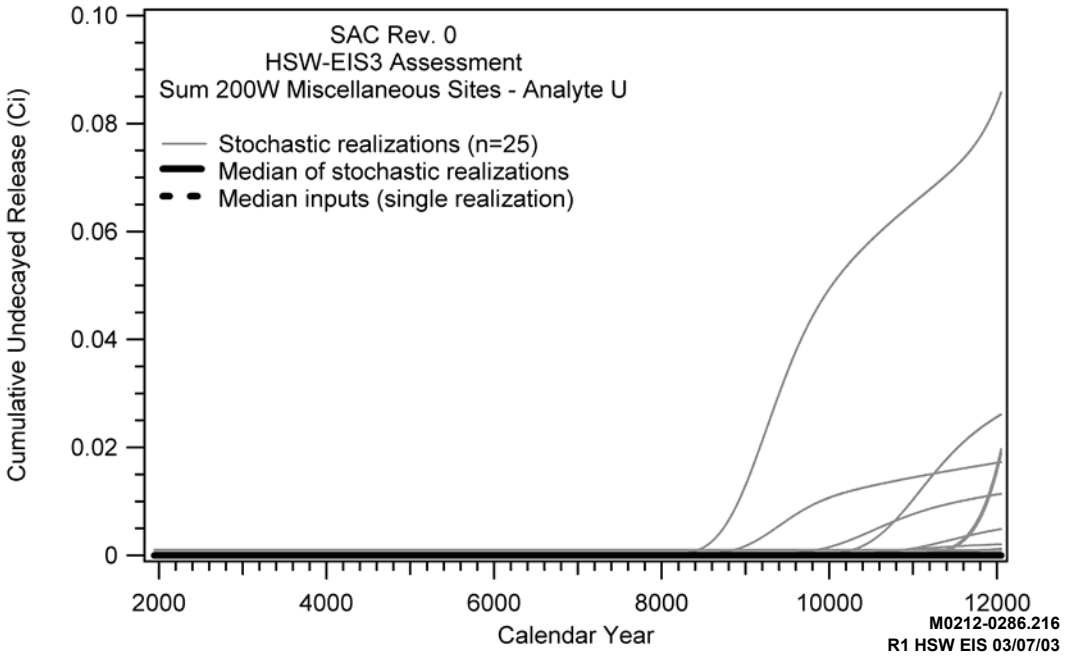

Figure L.36. SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Other Sites in the 200 West Area

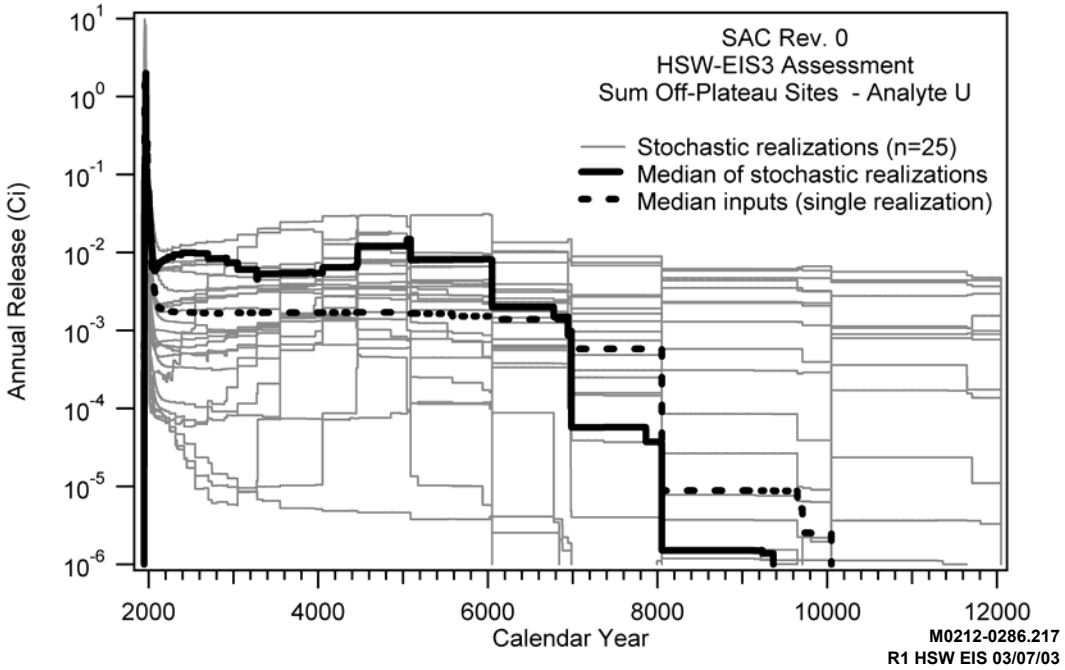

Figure L.37. SAC Results for Annual Vadose Zone Release of Uranium from All Other Sites Outside the 200 East and 200 West Areas

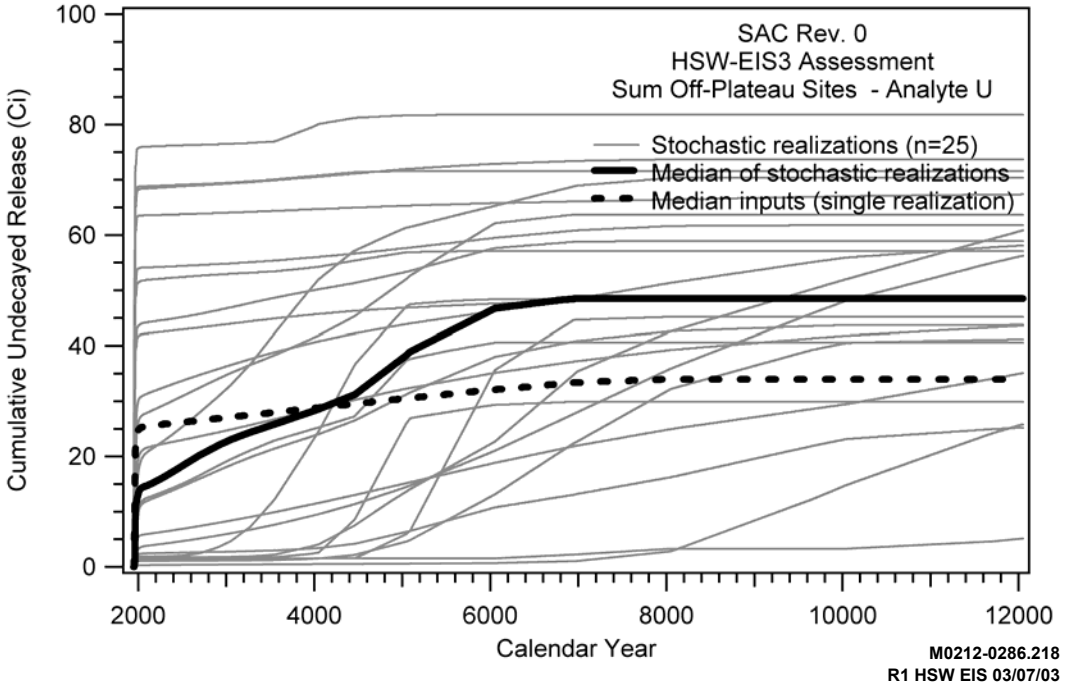

Figure L.38. SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Other Sites Outside the 200 East and 200 West Areas

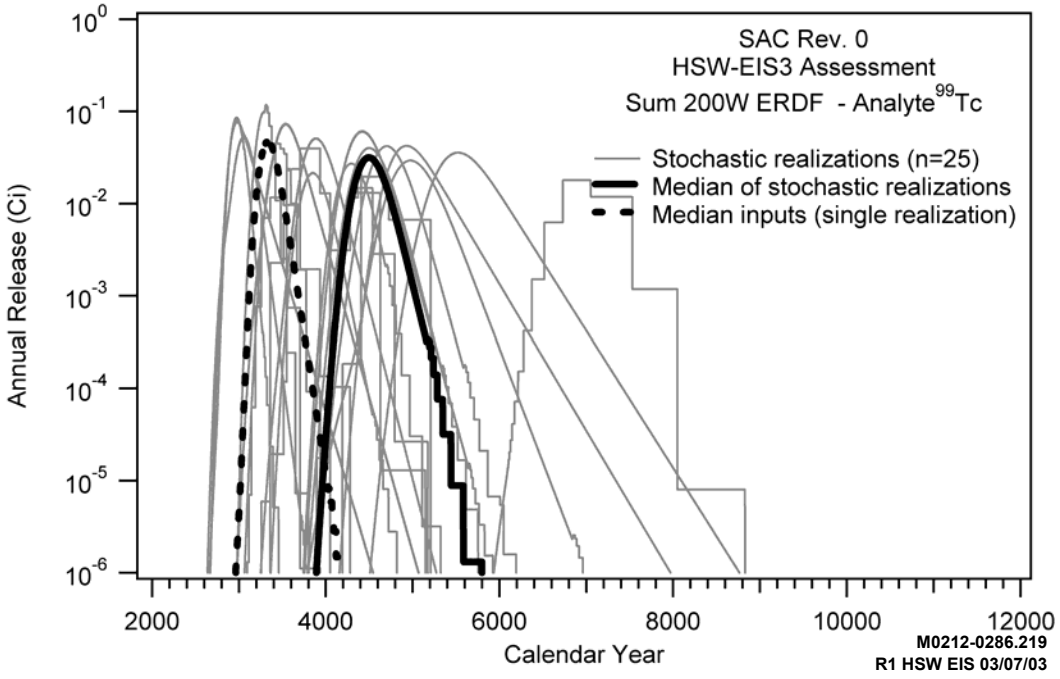

Figure L.39. SAC Results for Annual Vadose Zone Release of Technetium-99 from ERDF

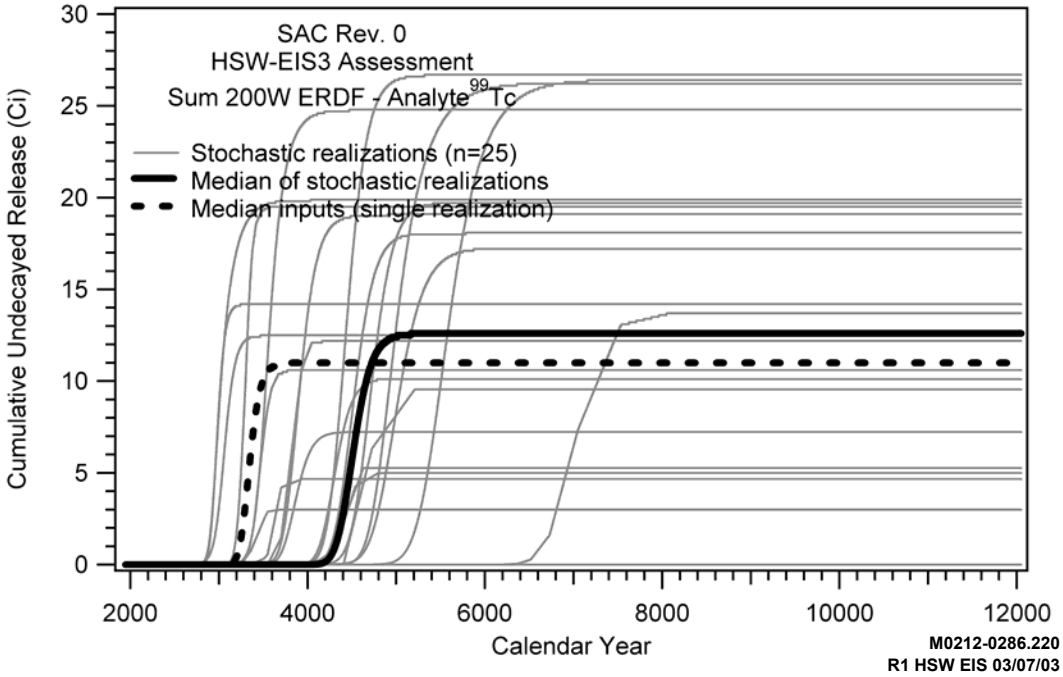

Figure L.40. SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from ERDF

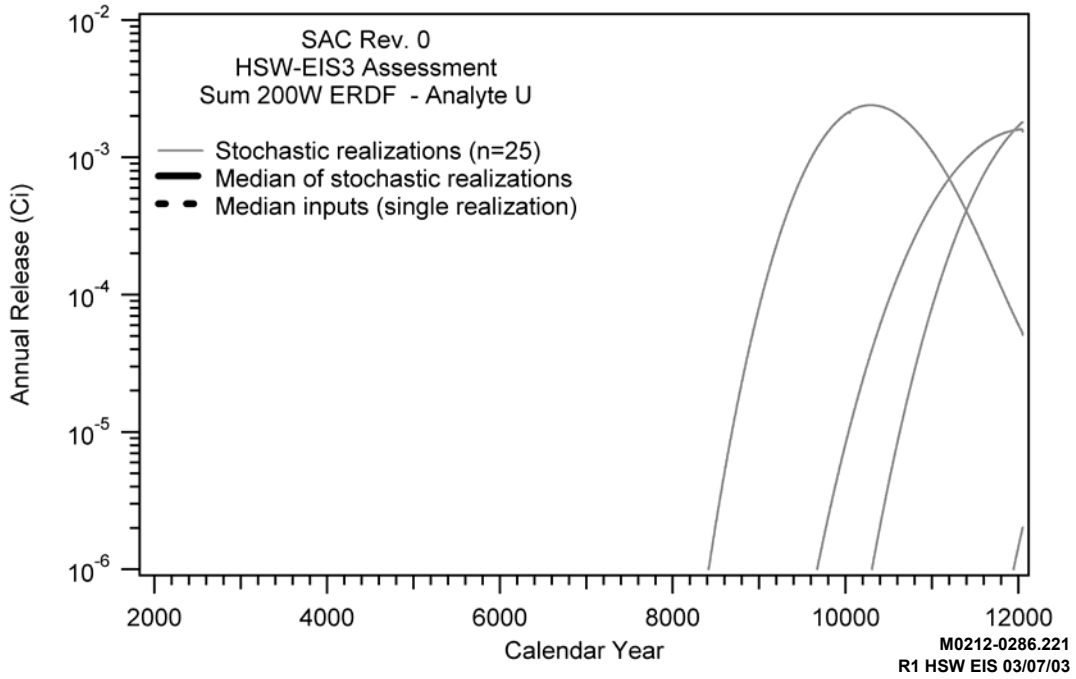

Figure L.41. SAC Results for Annual Vadose Zone Release of Uranium from the ERDF

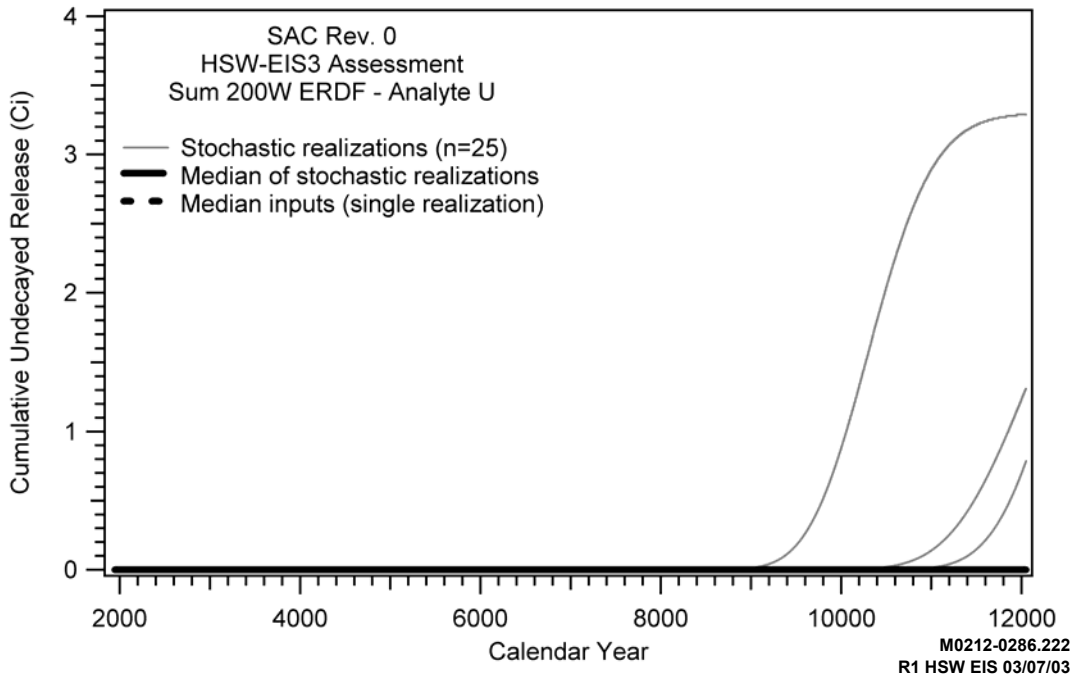

Figure L.42. SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from the ERDF

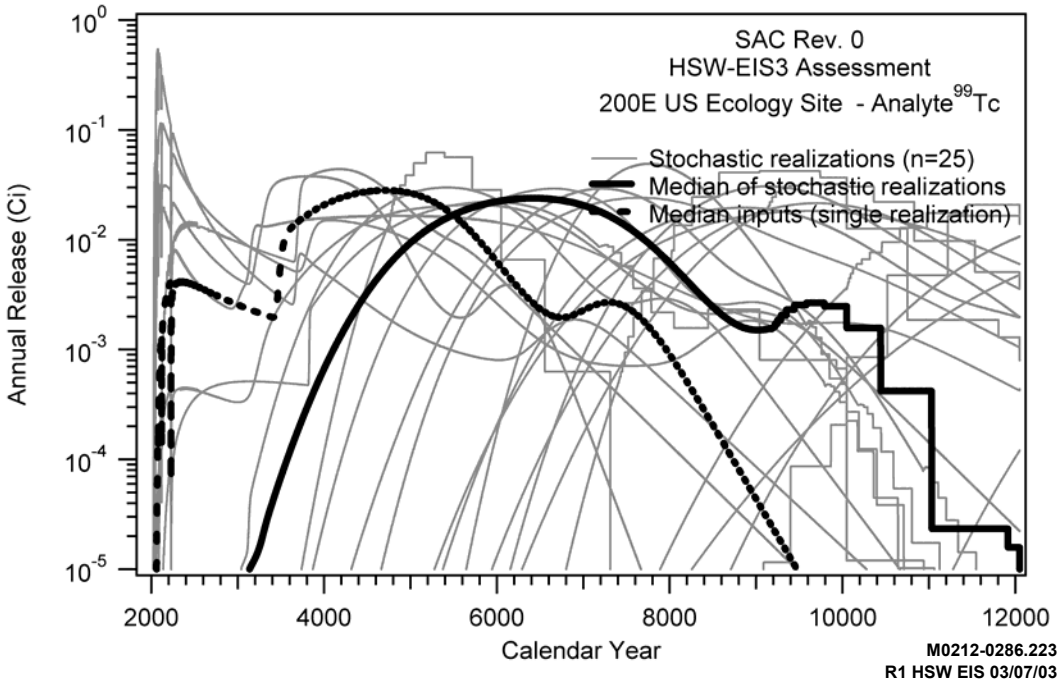

Figure L.43. SAC Results for Annual Vadose Zone Release of Technetium-99 from the Commercial Low-Level Radioactive Waste Disposal (US Ecology, Inc.) Site

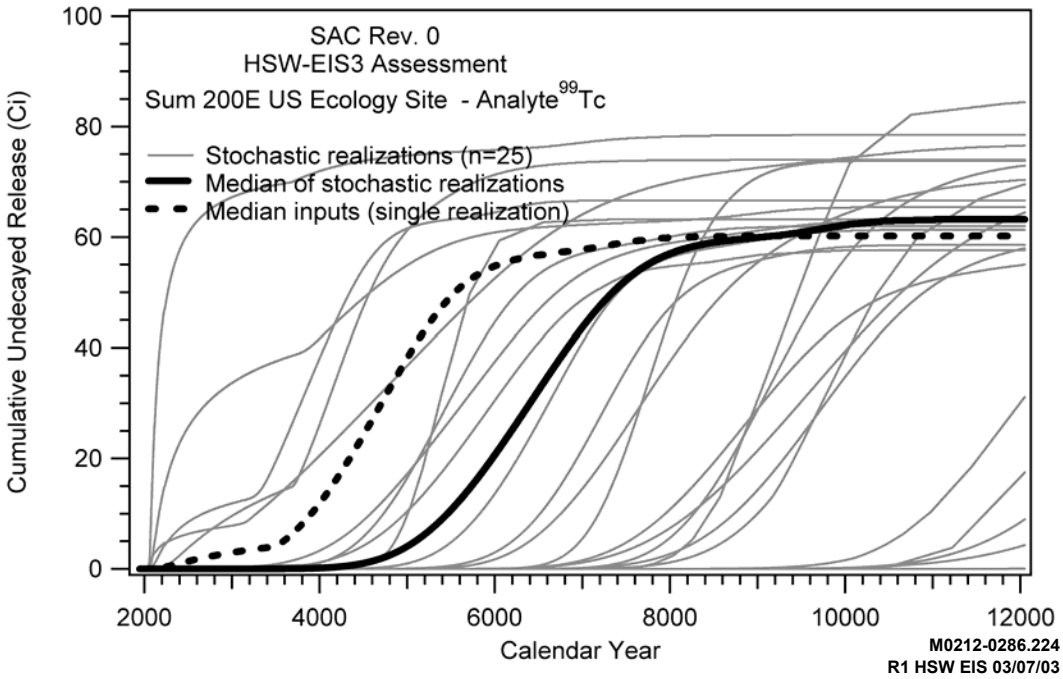

Figure L.44. SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from the Commercial Low-Level Radioactive Waste Disposal (US Ecology, Inc.) Site

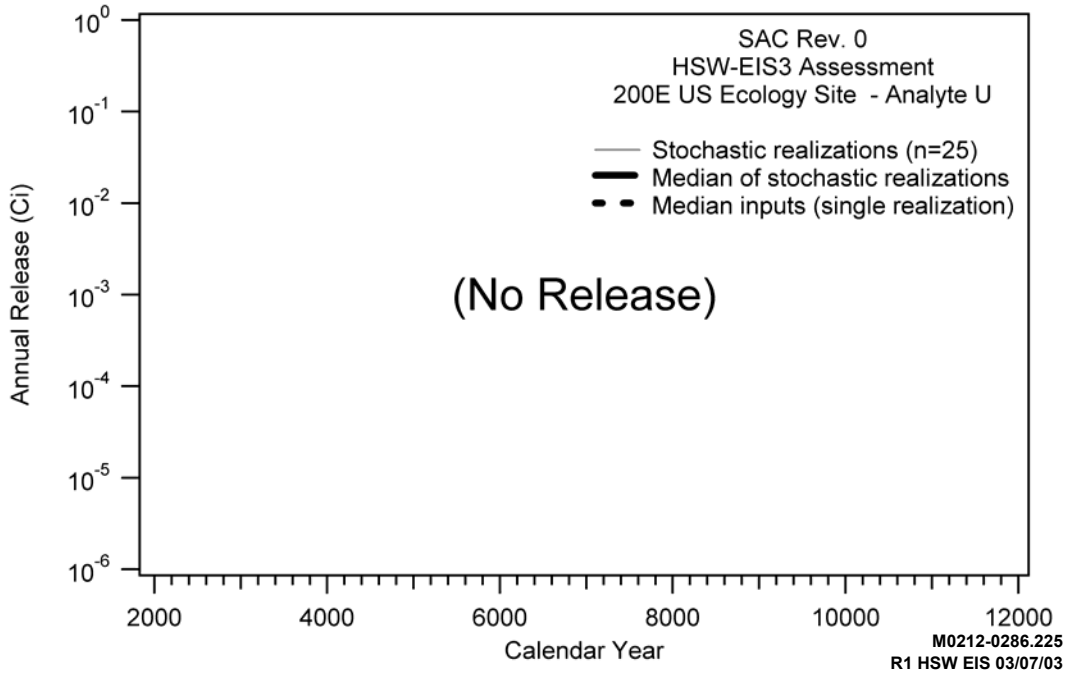

Figure L.45. SAC Results for Annual Vadose Zone Release of Uranium from the Commercial Low-Level Radioactive Waste Disposal (US Ecology, Inc.) Site

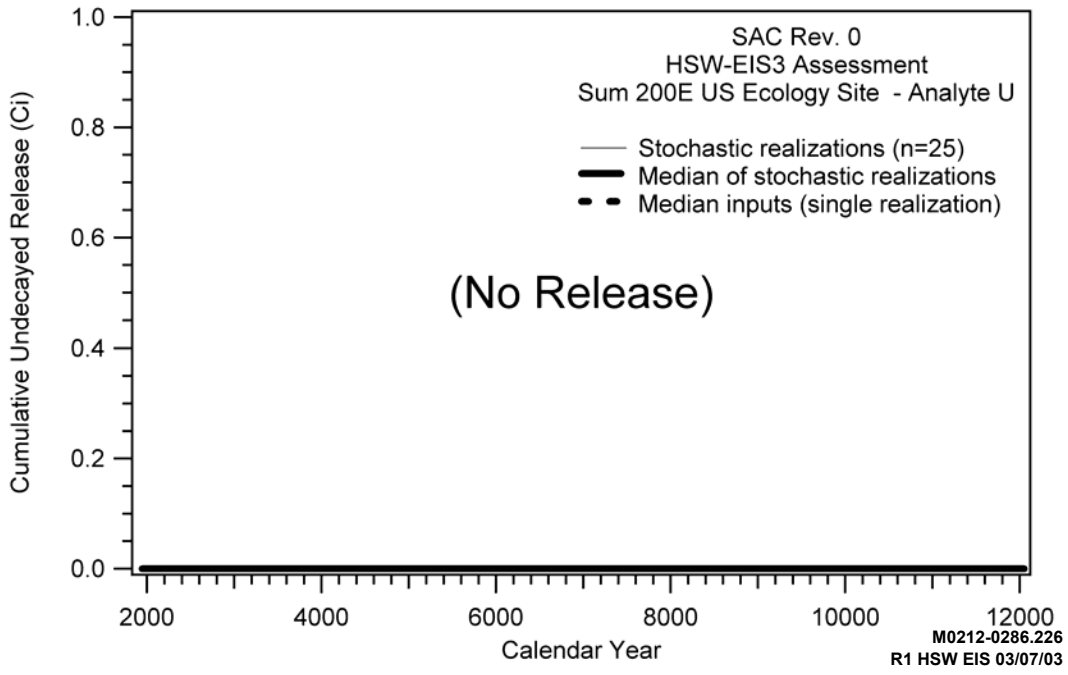

Figure L.46. SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from the Commercial Low-Level Radioactive Waste Disposal (US Ecology, Inc.) Site

L.3.2 Drinking Water Dose at Selected 200 East and 200 West Area Locations

Doses to humans calculated using the SAC software and data are summarized in this section. The exposure scenario has an adult human drinking 2 L per day of contaminated groundwater. The stochastic capability of SAC was employed for these simulations, so the following results are shown in each plot in this section:

- Individual stochastic results (25 realizations) are shown in black.
- The median result of the 25 realizations—that is, the realization that resulted in the median integrated cumulative dose in the year 12050 A.D. (at the end of the simulation)—is shown in blue.
- The median-inputs simulation—a separate single-realization simulation with SAC using the median value of all stochastic input variables—is shown in red.

The variability in the stochastic results is due to variability in the inventory, release, and transport of technetium-99 and uranium. The human dose calculations use fixed inputs.

The doses provided in this section are based on all waste at the Hanford Site except the ILAW, melters, and naval reactor compartments. Cumulative releases to groundwater for HSW excluding ILAW disposed of in the Central Plateau range from approximately 323 to approximately 445 Ci for technetium-99 during the 10,000-year analysis period. This compares with a range of release to groundwater between approximately 1530 and 2310 Ci of technetium-99 for all Hanford wastes except ILAW. The contribution from HSW excluding ILAW would amount to about 20 percent of the cumulative release from Hanford sources excluding ILAW. The median release of technetium-99 from HSW excluding ILAW was approximately 390 Ci while the median release for all Hanford sources except ILAW was approximately 2000 Ci. The ILAW cumulative release of technetium-99 for the base case (Mann et al. 2001) considering the full technetium-99 inventory was approximately 86 curies by the end of the 10,000-yr post-closure period. Accordingly the contribution from HSW including ILAW would amount to about 25 percent of the cumulative release from all Hanford sources after 10,000 years.

For uranium, the cumulative releases to groundwater for Hanford solid waste disposed of in the Central Plateau range from 0 to approximately 94 Ci. However of all realizations simulated, no realizations showed any release to groundwater from HSW in the 200 East Area, and only 5 of 25 realizations show any release of uranium to groundwater from HSW in the 200 West Area. Thus, in an average (or median) sense, HSW deposits would release no uranium to groundwater over the 10,000 yr period of analysis. This compares with a median release of approximately 84 Ci and a range of release to groundwater from the 25 realizations of between approximately 10 to 300 Ci of uranium for all Hanford wastes except ILAW. Of the five realizations of non-zero uranium release from HSW in the 200 West Area, the cumulative release ranged from 0 to approximately 90 Ci. The contribution from uranium in Hanford solid waste lies between 0 and 30 percent of the cumulative release from all Hanford sources. However, the median release of uranium from Hanford solid waste was zero while the median release for all Hanford sources (except ILAW) was approximately 84 Ci.

L.3.2.1 Drinking Water Dose at the Northeast Corner of the 200 West Area

The drinking water dose to a human from technetium-99 using groundwater approximately 1 km (0.62 mi) outside the northeast corner of 200 West Area is provided in Figure L.47. The location was chosen to represent the highest doses from the local groundwater plume. The drinking water dose to a human from uranium at the same location is provided in Figure L.48. Neither of these figures included ILAW waste-form impacts explicitly. However, ILAW disposal occurs in the 200 East Area, and existing and future groundwater flow will conduct plumes from ILAW release away from the 200 West Area location shown in these figures. The data for technetium-99 show peaks early and again after approximately 3000 years. Figure L.47 exhibits a peak dose from technetium-99 in the range of 1 to 3 mrem/yr and a median of less than 2 mrem/yr with much lower consequences in the 7000 to 10,000-year time frame (that is, a range of 0.001 to 0.01 mrem/yr and a median less than 0.002 mrem/yr). Figure L.48 exhibits a peak dose from uranium (that is, a range of 0.01 to 0.3 mrem/yr and a median of approximately 0.05 mrem/yr) and considerable variability in later years because of the sorption model for uranium (that is, a range of 0.0001 to 7 mrem/yr and a median of approximately 0.04 mrem/yr).

L.3.2.2 Drinking Water Dose at the Southeast Corner of the 200 East Area

The drinking water dose to a human from technetium-99 using groundwater from approximately 1 km (0.62 mi) outside the southeast corner of 200 East Area is provided in Figure L.49. The location was chosen to represent the highest doses from the local groundwater plume. The drinking water dose to a human from uranium at the same location is provided in Figure L.50. Neither of these figures includes ILAW waste-from impacts. The technetium-99 results show peaks early and again throughout the 10,000-year period. Figure 5.49 exhibits a peak median dose from technetium-99 in the range of 0.3 to 2 mrem/yr during the 10,000-year period. Peaks within all realizations range to 100 mrem/yr. Figure 5.50 exhibits a peak median dose from uranium of less than 1 mrem/yr early with a long-term median value of less than 0.01 mrem/yr. There is considerable variability in later years because of the sorption model for uranium (that is, after 10,000 years, there is a range of approximately 0.001 to 1 mrem/yr, but the median is less than 0/01 mrem/yr).

L.3.2.3 Drinking Water Dose at the Northwest Corner of the 200 East Area

The drinking water dose to a human from technetium-99 using groundwater from approximately 1km (0.62 mi) outside the northwest corner of 200 East Area is provided in Figure L.51. The location was chosen to represent the highest doses from the local groundwater plume. The drinking water dose to a human from uranium at the same location is provided in Figure L.52. These figures exclude the influence of the ILAW waste-form impact. The technetium-99 results show peaks early and again throughout the 10,000-year period. Figure L.51 exhibits a peak median dose from technetium-99 in the range of 0.1 to 3 mrem/yr during the 10,000-year period. Figure 5.52 exhibits a peak median dose from uranium of approximately 3 mrem/yr with a long-term median value of less than 0.01 mrem/yr. There is considerable variability in later years because of the sorption model for uranium (that is, after 10,000 years, there is range of approximately 0.001 to 1 mrem/yr, but the median is less than 0.01 mrem/yr).

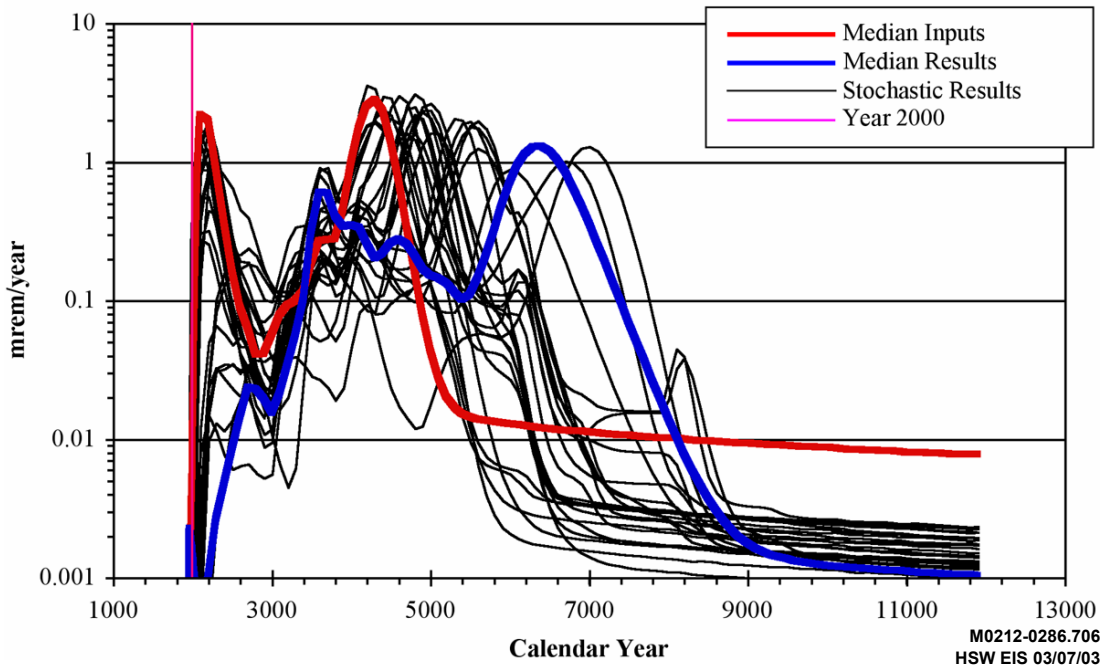

Figure L.47. Annual Drinking Water Dose from Technetium-99 in Groundwater 1 Kilometer Northeast of the 200 West Area

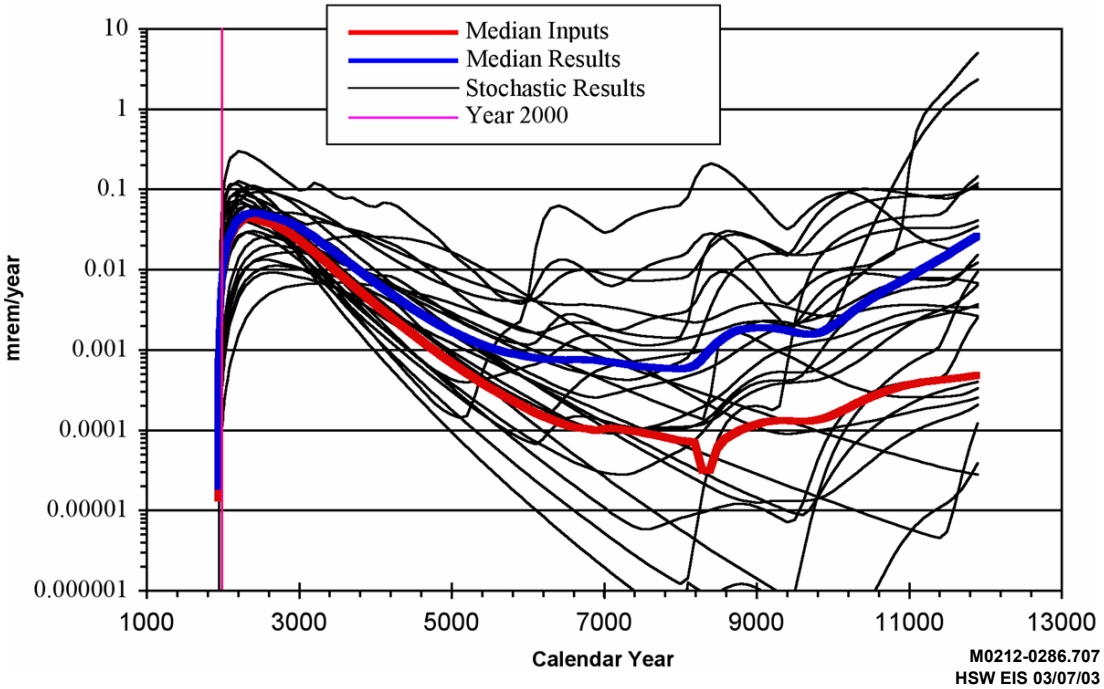

Figure L.48. Annual Drinking Water Dose from Uranium in Groundwater 1 Kilometer Northeast of the 200 West Area

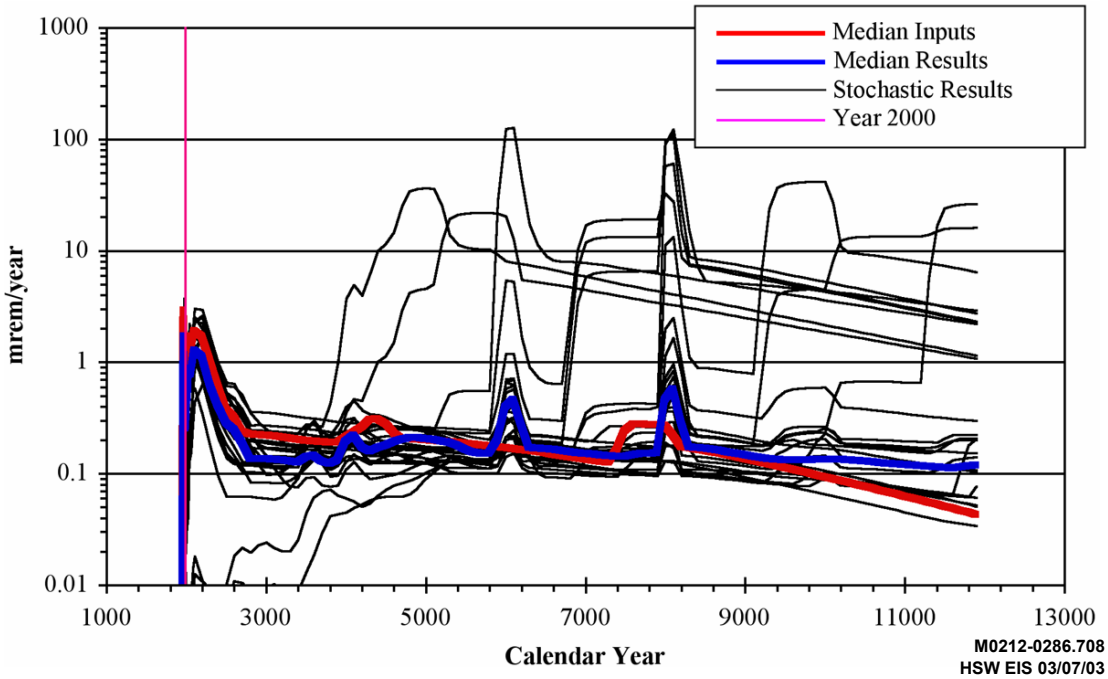

Figure L.49. Drinking Water Dose from Technetium-99 in Groundwater 1 Kilometer Southeast of the 200 East Area from All Hanford Sources Except ILAW, Melters, and Naval Reactors

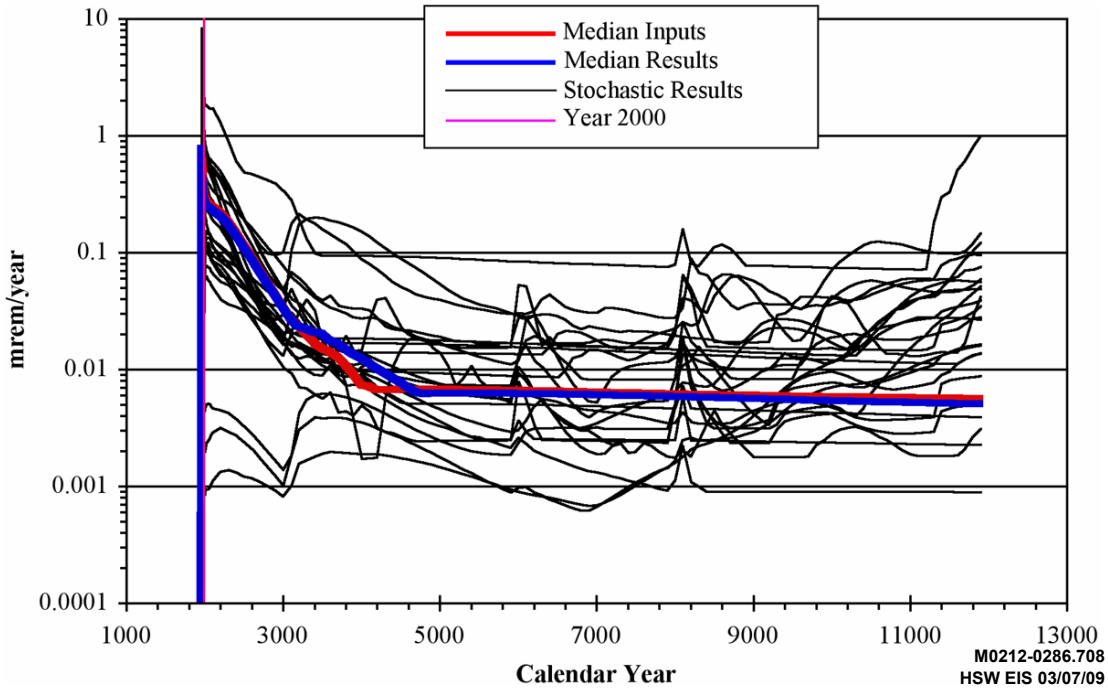

Figure L.50. Drinking Water Dose from Uranium in Groundwater 1 Kilometer Southeast of the 200 East Area from All Hanford Sources Except ILAW, Melters, and Naval Reactors

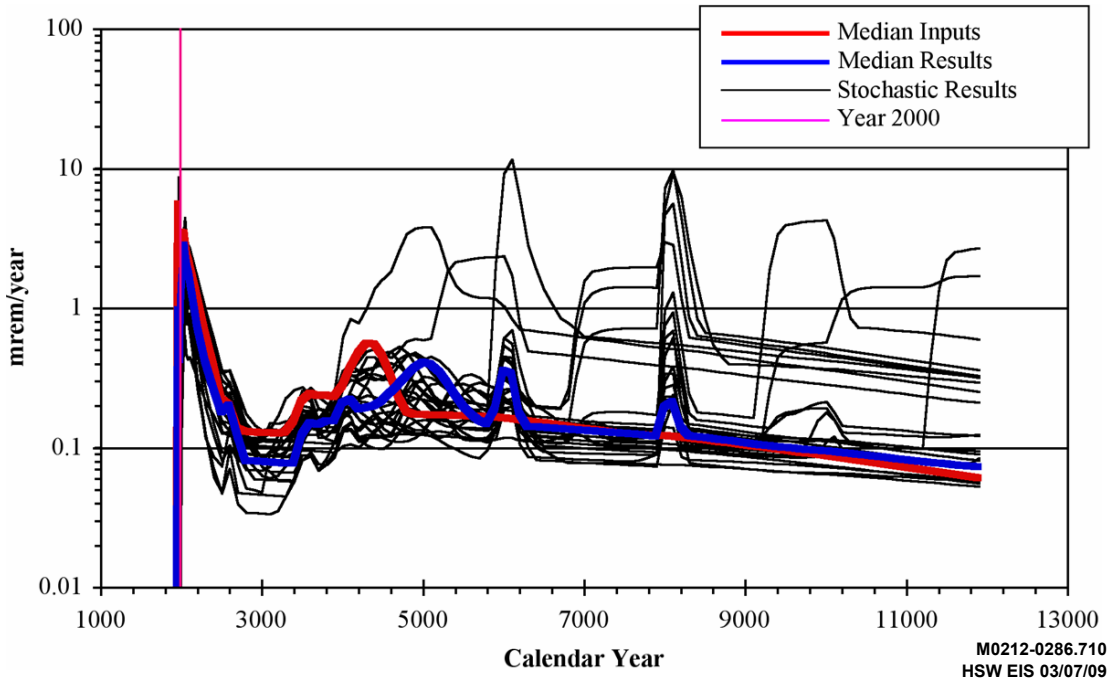

Figure L.51. Drinking Water Dose from Technetium-99 in Groundwater 1 Kilometer Northwest of the 200 East Area from All Hanford Sources Except ILAW, Melters, and Naval Reactors

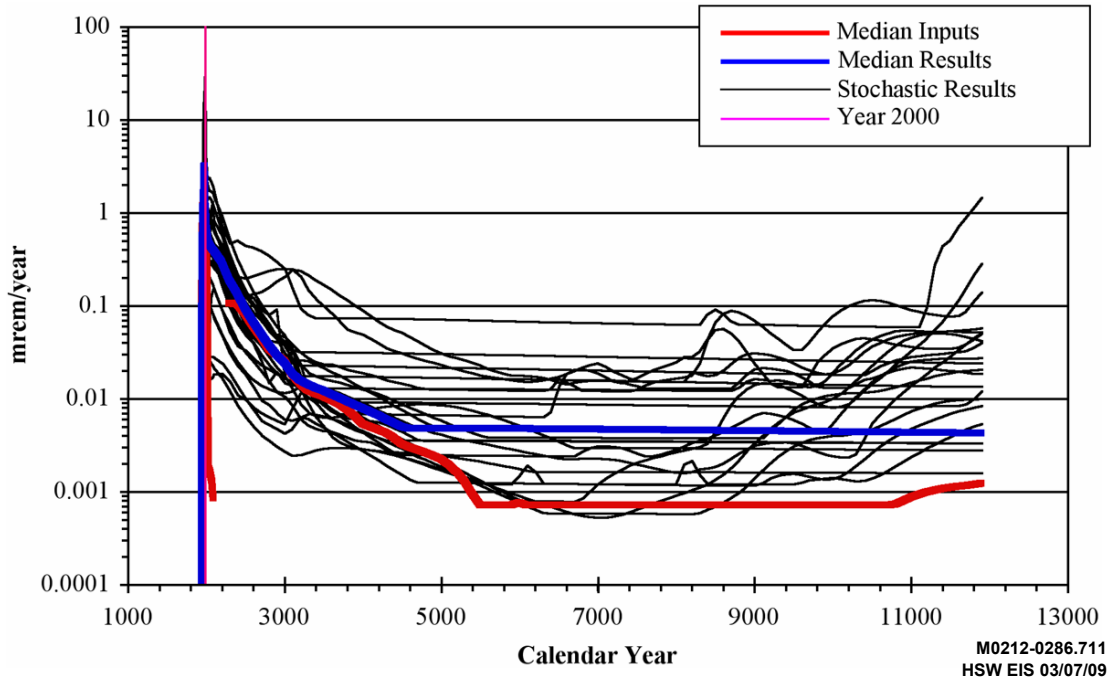

Figure L.52. Drinking Water Dose from Uranium in Groundwater 1 kilometer Northwest of the 200 East Area from All Hanford Sources Except ILAW, Melters, and Naval Reactors

L.3.3 Dose from Columbia River Water at the City of Richland Pumping Station

Annual dose to humans based on consumption of river water is summarized in this section. The exposure scenario has an adult human drinking 2 liters per day of contaminated river water from the modeled near-shore point nearest the City of Richland Pumping Station. The stochastic capability of SAC was employed for these simulations, so the following results are shown in each plot in this section:

- Individual stochastic results (25 realizations) are shown in black.

- The median result of the 25 realizations—that is, the realization that resulted in the median integrated cumulative dose in the year 9900 A.D.—is shown in blue. Although the groundwater simulations continued through the year 12050 A.D., the river simulations were terminated at the year 9900 A.D. due to software design constraints.

- The median-inputs simulation—a separate single-realization simulation with SAC using the median value of all stochastic input variables—is shown in red.

The variability in the stochastic results is due to the inventory, release, and transport of technetium-99 and uranium. The human dose model uses fixed inputs in the calculations. The doses provided in this section are based on all waste at the Hanford site and do not include background concentrations in the river. Thus, the doses are due entirely to Hanford contaminants, with most of the dose due to waste forms other than solid wastes.

L.3.3.1 Drinking Water Dose at the City of Richland Pumping Station

The drinking water dose to a human from technetium-99 using water concentrations calculated near the City of Richland Pumping Station is provided in Figure L.53. This location is downriver from all groundwater plumes of Hanford origin. The maximum estimated annual dose from technetium-99 over all realizations from the year 2000 through 9900 A.D. is less than 0.00008 mrem/yr, while the peak median dose was approximately 0.00004 mrem/yr. The annual drinking water dose to a human from uranium at the same location is provided in Figure L.54. The maximum annual dose from uranium over all realizations from the year 2000 through 12050 A.D. is less than 0.002 mrem/yr, while the peak median dose was approximately 0.00005 mrem/yr.

L.3.4 Annual Drinking Water Dose at Selected 200 East Area and Columbia River Locations from Hanford Sources Including ILAW

The deterministic capability of SAC was employed with results of the ILAW performance assessment (Mann et al. 2001), which were scaled to current inventory estimates to provide an initial estimate of the cumulative impact of all Hanford sources including ILAW. These deterministic results portray the median-inputs case of the initial assessment using SAC and the base case of the ILAW performance assessment (Mann et al. 2001). Essentially, the 2 L/d dose impacts from the ILAW inventories of technetium-99 and uranium reported in the ILAW performance assessment (Mann et al. 2001) are

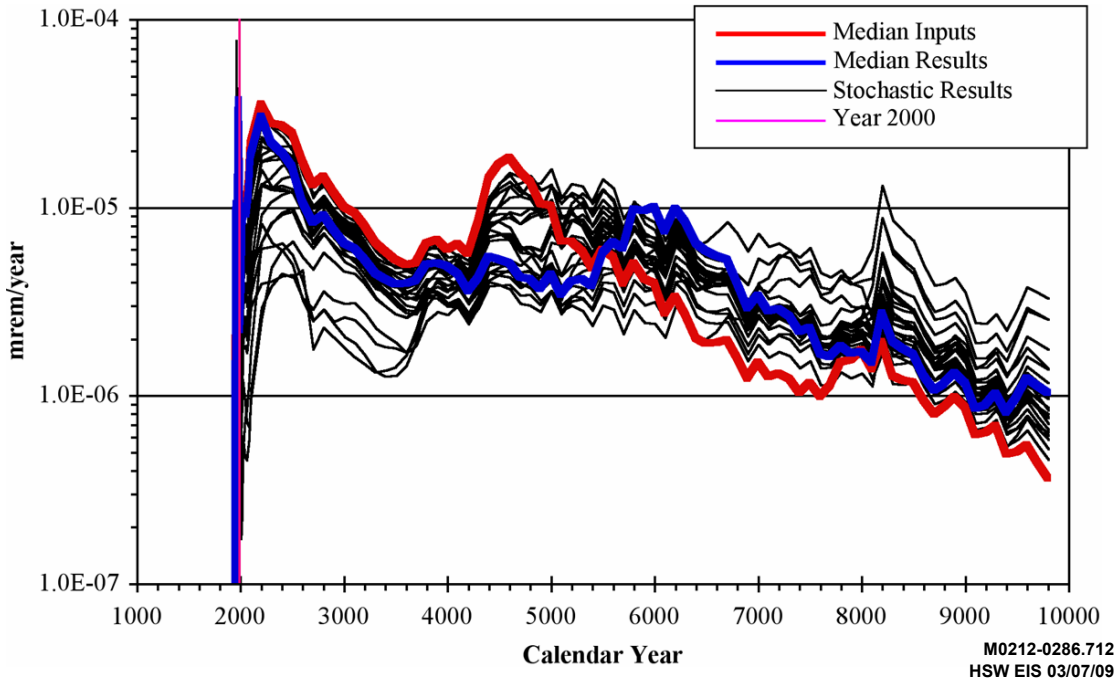

Figure L.53. Drinking Water Dose at the City of Richland Pumping Station from Technetium-99 Due to All Hanford Sources Except ILAW, Melters, and Naval Reactors

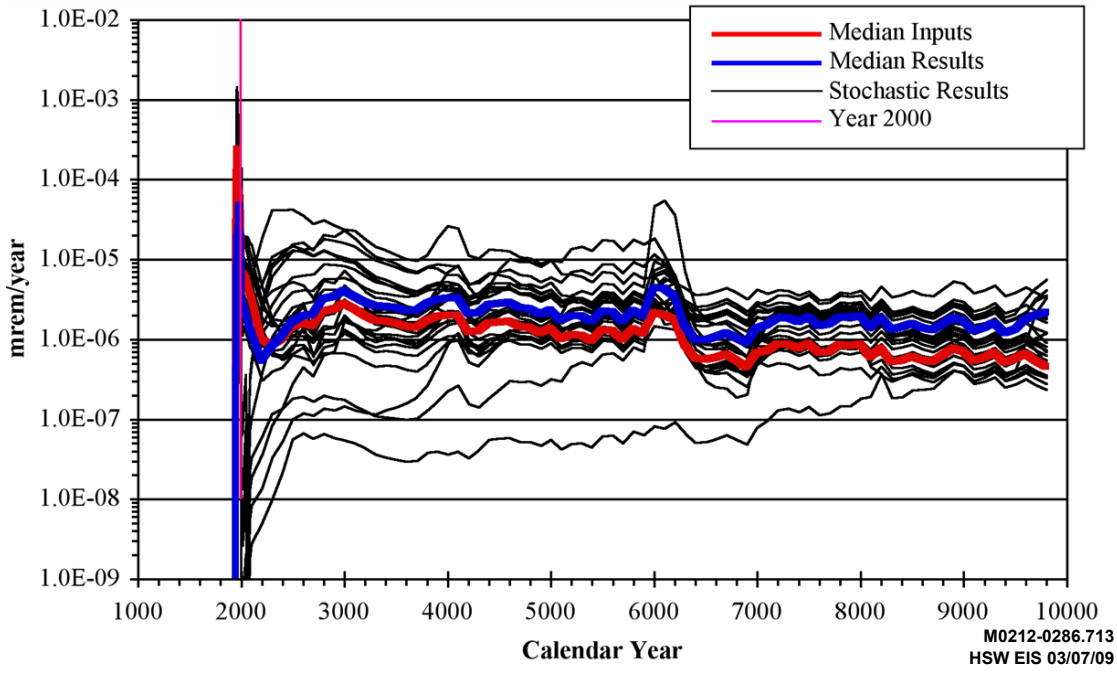

Figure L.54. Drinking Water Dose at the City of Richland Pumping Station from Uranium Due to All Hanford Sources Except ILAW, Melters, and Naval Reactors

superimposed on the SAC median-value simulation. A series of three plots show combined SAC and ILAW results at a point 1-km southeast of the 200 East Area and at a point of analysis near the shore of the Columbia River at the City of Richland Pumping Station.

The cumulative impact for all Hanford sources is provided in Figure L.55. This is the annual drinking water dose from a 2 L/d drinking water scenario for technetium-99 at a point of analysis approximately 1 km (0.62 mi) southeast of the 200 East Area. The curve is a composite of the SAC initial assessment result and the base case ILAW result (Mann et al. 2001). To account for the current estimate of 25,500 Ci of technetium-99 in low-activity waste from the single- and double-shell tanks, the ILAW analysis of a 5790 Ci technetium-99 source has been scaled accordingly.

The cumulative result shown in Figure L.55 exhibits an initial peak prior to the year 2000 and a secondary peak in the next two centuries. The secondary peak is approximately 1 mrem/yr and is related to releases from liquid discharge sites in the 200 East Area. Additional, but lower, secondary peaks, 0.03 mrem/yr, appear in approximately 4300 A.D. and 7500 A.D. Releases from solid waste disposal facilities in the 200 West Area are responsible for the earlier of these two secondary peaks. Tank waste residuals releasing from the 200 East Area, modeled as 1 percent residual tank waste volume in a salt cake waste form, are responsible for the last secondary peak.

By the end of the 10,000-year, post-closure period, the cumulative dose from all Hanford sources is approximately 0.06 mrem/yr, of which approximately 0.02 mrem/yr is from ILAW and 0.04 mrem/yr is from all other Hanford sources. Based on uncertainty in the groundwater conceptual model, the ILAW contribution may be four times larger. Thus, the ILAW contribution may be 0.08 mrem/yr and may be comparable to or larger than that for all other Hanford sources. For this alternate conceptual model, the cumulative 2-L/d dose would be approximately 0.12 mrem/yr at 10,000 years post-closure. Note that ILAW release and associated dose impacts play a role in the last several thousand years, and do not substantially alter the secondary peaks described earlier.

A comparison of consequences from consuming 2 L/d of river water with and without the ILAW release of technetium-99 and uranium are provided in Figures L.56 and L.57 for the Columbia River at the City of Richland Pumping Station. Results from the SAC median-input case of the initial assessment and from the ILAW performance assessment base case are shown on each figure. Figure L.56 shows that dose originating from the low-activity waste source containing 25,500 Ci of technetium-99 is approximately equivalent to or slightly greater than the dose originating from all other Hanford wastes. The cumulative dise is 1.0×10^{-6} mrem/yr at 10,000 years post-closure, and this result is five orders-of-magnitude below the dose predicted at the 200 East area location.

The comparison graphic of consequences from uranium is provided in Figure L.57. After 10,000 years post-closure and at the time of greatest ILAW uranium impact, the dose from uranium is estimated to be approximately an order-of-magnitude below that of all other Hanford sources. Combined, the estimated dose is less than 1.0×10^{-6} mrem/yr.

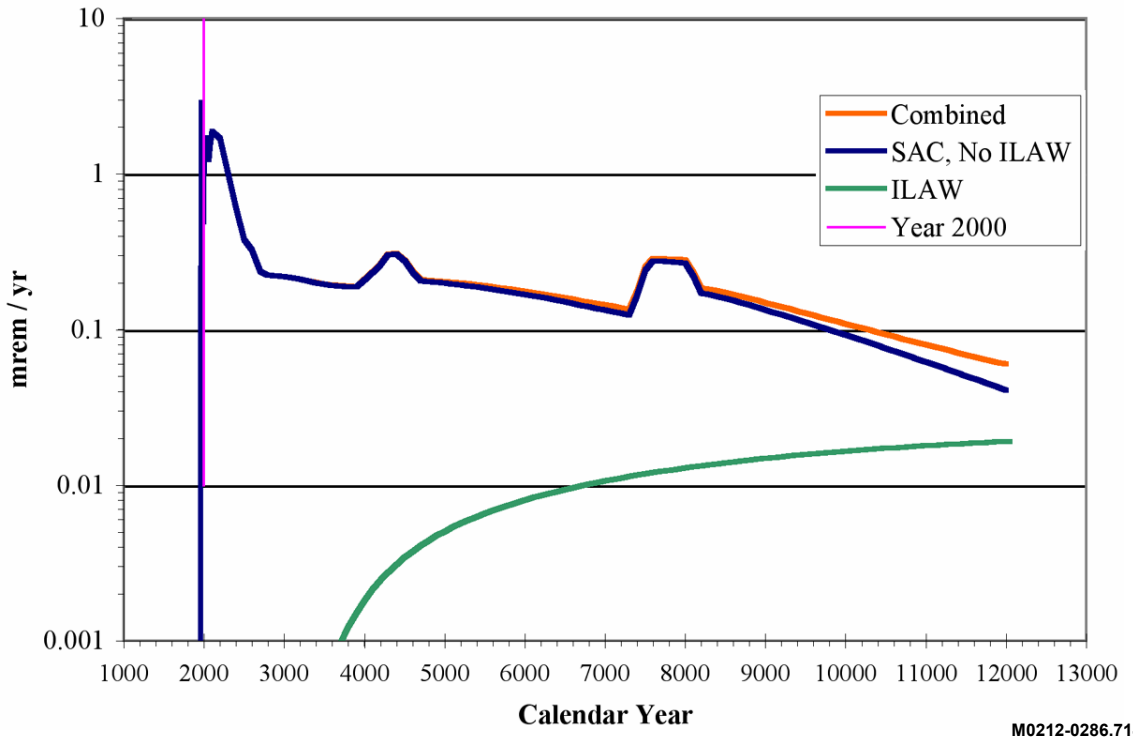

Figure L.55. Annual Drinking Water Dose from Technetium-99 in Groundwater 1 Kilometer Southeast of the 200 East Area from Hanford Sources Including ILAW

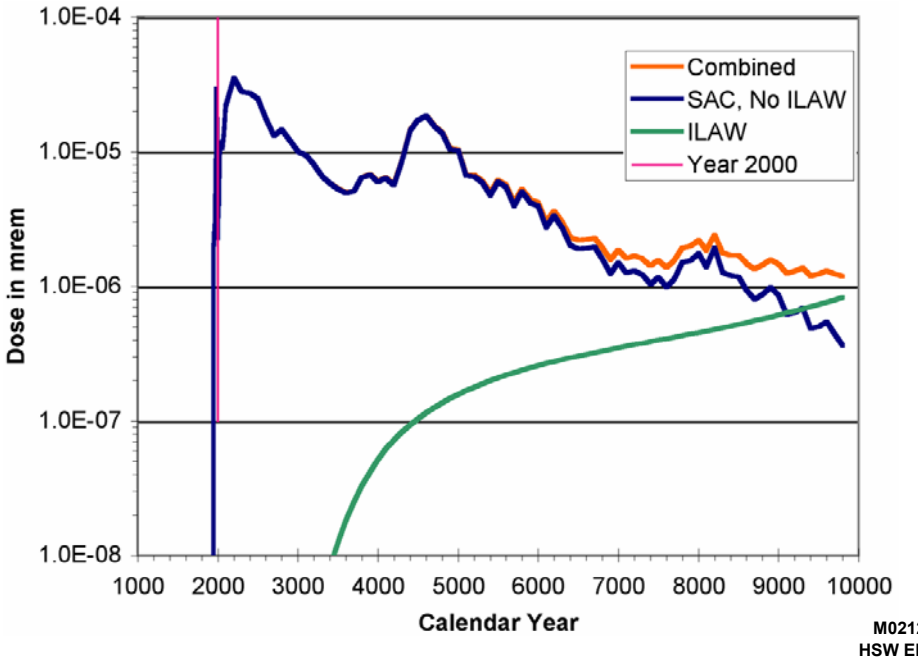

Figure L.56. Annual Drinking Water Dose from Technetium-99 in the Columbia River at the City of Richland Pumping Station from Hanford Sources Including ILAW

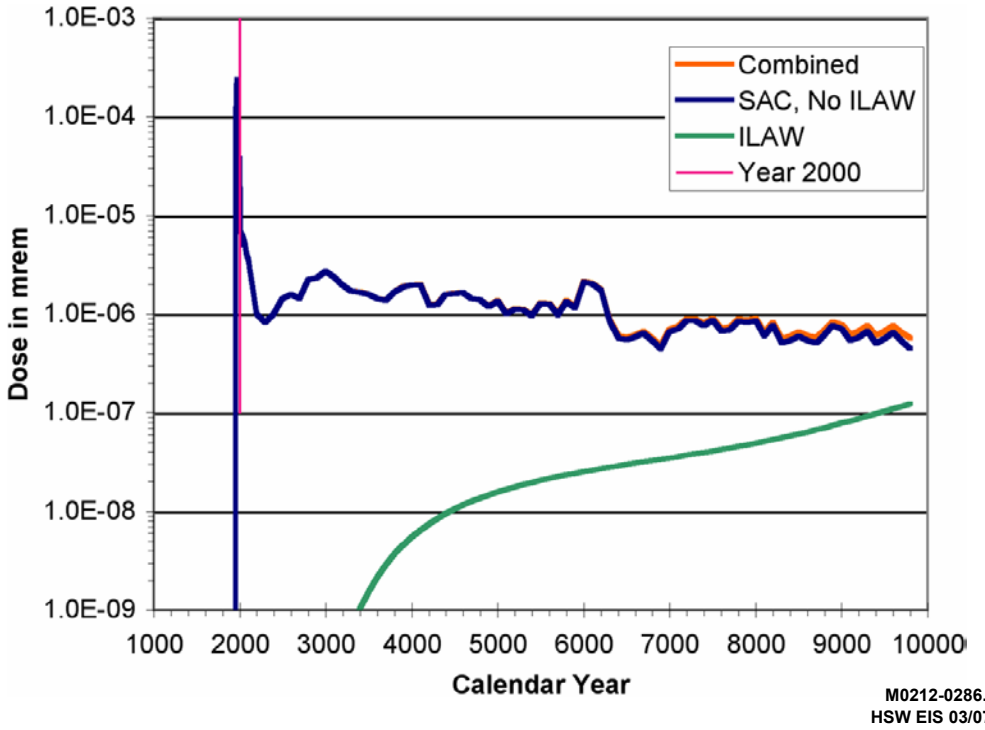

Figure L.57. Annual Drinking Water Dose from Uranium in the Columbia River at the City of Richland Pumping Station from Hanford Sources Including ILAW

The dose from technetium-99 at the City of Richland (Figure L-61) exhibits the secondary peak structure seen in the dose from technetium-99 near the 200 East Area. However, the dose from consumption of river water exhibits a greater variability in both Figures L.56 and L.57 because of the underlying variability associated with Columbia River discharge. Secondary peak structure is greatly subdued in the dose from uranium plot (Figure L.57) because uranium is sorbed onto subsurface sediments and river sediments.

The results are an approximation achieved by superimposing the results of two independently conducted analyses. Nevertheless, the results indicate that the contribution from ILAW, which represents a substantial fraction of the technetium-99 inventory at Hanford, while being equivalent to the initial assessment results does not substantially influence the overall dose prediction made in the initial assessment for all wastes other than ILAW.

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